

Technical Support of Standards for High-Level
Radioactive Waste Management. Volume C
Migration Pathways

Arthur D. Little, Inc.
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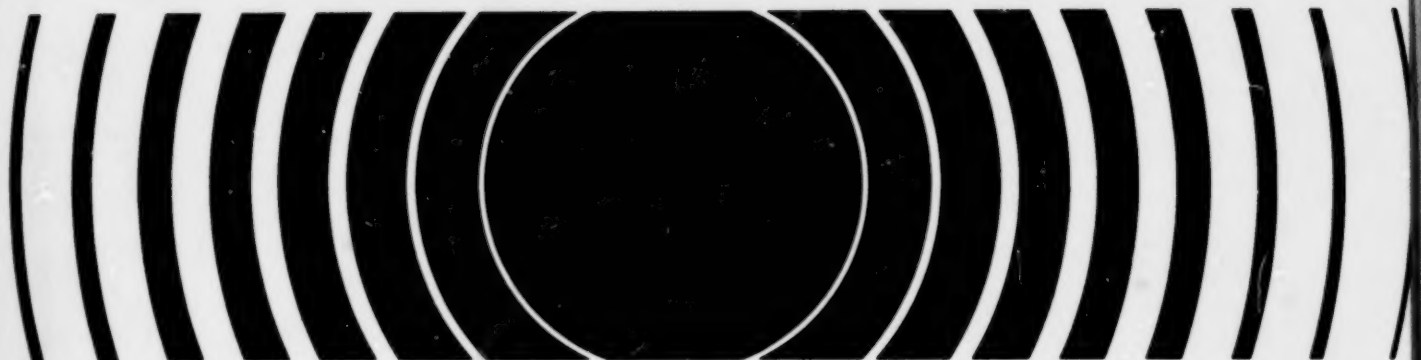
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Volume C Migration Pathways



TECHNICAL SUPPORT OF STANDARDS FOR
HIGH-LEVEL RADIOACTIVE WASTE MANAGEMENT

TASK C REPORT

ASSESSMENT OF MIGRATION PATHWAYS

EPA Contract No. 68-01-4470

Prepared by

Arthur D. Little, Inc.
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March-July 1977

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FOREWORD

A major Federal effort is underway to develop methods for disposal of high-level radioactive waste in deep geologic repositories. An important element of this program is the development and promulgation by the U.S. Environmental Protection Agency (EPA) of environmental standards for the management of these wastes.

In anticipation of its efforts to develop these standards, EPA recognized that it would be necessary to estimate the expected and potential environmental impacts from potential geologic repositories using modeling techniques based upon as thorough an understanding as possible of the uncertainties involved in the quantities and characteristics of the wastes to be managed, the effectiveness of engineering controls, and the potential migration and accidental pathways that might result in radioactive materials entering the biosphere. Consequently, in March 1977, the EPA contracted with Arthur D. Little, Inc., for a study to provide technical support for its development of environmental regulations for high-level radioactive wastes. This study was divided into the following four tasks:

Task A - Source Term Characterization/Definition

Task B - Effectiveness of Engineering Controls

Task C - Assessment of Migration Pathways

Task D - Assessment of Accidental Pathways

The information presented in the reports on these tasks was developed principally during the period March 1977 to February 1978. In the case of this report, Task C, the information contained in it was prepared during the period March-July 1977. There are many national and international programs underway to develop additional data, especially in the fields of waste forms, knowledge of geology and geohydrology, and risk assessment. The information presented in these reports has been developed on conceptual bases and is not intended to be specific to particular conditions at geologic repositories.

TABLE OF CONTENTS

	<u>Page</u>
Acknowledgements	111
Foreword	iv
List of Tables	viii
List of Figures	x
C-1.0 <u>INTRODUCTION</u>	1
C-2.0 <u>SUMMARY AND CONCLUSIONS</u>	2
2.1 ASSESSMENT OF GEOLOGIC SITE SELECTION FACTORS	2
2.2 ANALYSIS OF MIGRATION POTENTIAL	4
2.3 DOSE-TO-MAN CONSIDERATIONS	5
C-3.0 <u>ASSESSMENT OF GEOLOGIC SITE SELECTION FACTORS</u>	8
3.1 IMPORTANT INTRINSIC PROPERTIES OF GEOLOGIC MEDIA	10
3.1.1 <u>Hydrogeologic Properties</u>	11
3.1.2 <u>Geochemical Properties</u>	16
3.1.3 <u>Thermal Properties</u>	27
3.1.4 <u>Engineering Properties</u>	28
3.2 IDENTIFICATION AND DISCUSSION OF SITE SELECTION FACTORS	28
3.2.1 <u>Lithology</u>	30
3.2.2 <u>Stratigraphy</u>	36
3.2.3 <u>Structural Geology</u>	39
3.2.4 <u>Hydrogeology</u>	43
3.2.5 <u>Tectonics</u>	48
3.2.6 <u>Erosion and Denudation</u>	54
3.2.7 <u>Mineral Resources</u>	55
3.3 CATEGORIZATION AND RANKING OF SITE SELECTION FACTORS	56
3.3.1 <u>Categorization of Factors</u>	57
3.3.2 <u>Ranking of Factors</u>	59
3.4 CONCEPTUAL GEOLOGIC MODELS	60
3.4.1 <u>Bedded Salt</u>	61

TABLE OF CONTENTS (cont.)

	<u>Page</u>
3.4.2 <u>Salt Domes</u>	64
3.4.3 <u>Shale</u>	66
3.4.4 <u>Igneous Rocks (Basalt Model)</u>	68
C-4.0 <u>ANALYSIS OF MIGRATION POTENTIAL</u>	71
4.1 <u>MODES OF WATER CONTACT</u>	71
4.2 <u>EVALUATION OF LEACH RATES</u>	78
4.2.1 <u>Introduction</u>	78
4.2.2 <u>Empirical Leach Models</u>	79
4.2.3 <u>Theoretical Leach Models</u>	82
4.3 <u>GEOSPHERE TRANSPORT THEORY ASSESSMENT</u>	83
4.3.1 <u>Hydrology - Radionuclide Transport Model Review</u>	85
4.4 <u>MIGRATION OF RADIONUCLIDES IN THE GEOSPHERE</u>	90
4.4.1 <u>Introduction</u>	90
4.4.2 <u>Results Using Analytical Model</u>	95
4.4.3 <u>Results Using Numerical Model</u>	104
4.5 <u>COMPARISON OF THE EFFECT OF LOW-GRADE URANIUM ORE VS. AN HLW REPOSITORY</u>	125
C-5.0 <u>DOSE-TO-MAN CONSIDERATIONS</u>	128
5.1 <u>OBJECTIVES AND OVERVIEW</u>	128
5.1.1 <u>Introduction and Purpose</u>	128
5.1.2 <u>Options and Considerations for Making Radiation Dose Assessments</u>	129
5.1.3 <u>Conditions of Exposure and Persons or Populations at Risk</u>	132
5.1.4 <u>Implications of the Use of a Linear-Nonthreshold Dose-Response</u>	136
5.2 <u>BASIS FOR DOSE-TO-MAN CALCULATIONS</u>	137
5.2.1 <u>Introduction</u>	137
5.2.2 <u>Repository Release Scenarios</u>	138
5.2.3 <u>Pathway Models</u>	142

TABLE OF CONTENTS (cont.)

	<u>Page</u>
5.2.4 <u>Exposure Routes</u>	150
5.3 RESULTS OF RADIATION DOSE CALCULATIONS	157
5.3.1 <u>Introduction</u>	157
5.3.2 <u>"Maximum" Individual Radiation Dose</u>	158
5.3.3 <u>"Maximum" Population Radiation Dose</u>	162
5.3.4 <u>"Reasonable" Population Dose</u>	164
5.4 EVALUATION OF DOSE-TO-MAN CONSIDERATIONS	166
5.4.1 <u>Worldwide Impact of Iodine-129</u>	166
5.4.2 <u>Comparison with Natural Background Radiation</u>	168
5.4.3 <u>Comparison with Radium-226 Currently Present in the Colorado River</u>	170
References Cited	173
 APPENDICES	
Appendix C-I Glossary	C-I-1
Appendix C-II Radioactive Waste Migration Model	C-II-1
Appendix C-III Parameters of Study	C-III-1
Appendix C-IV "Maximum" Individual and Population Dose Commitment Sample Calculations	C-IV-1
Appendix C-V "Reasonable" Population Dose Methodology and Sample Calculation	C-V-1

LIST OF TABLES

<u>Table No.</u>		<u>Page</u>
C-1	Example Intrinsic Properties for Typical Repository and Associated Lithologies	12
C-2	Permeability of Various Rock and Soil Types	15
C-3	Estimated Distribution Coefficients (K_d) and Retardation Factors (R_d) in a Typical Desert Soil	19
C-4	Distribution Coefficients (K_d) for Different Soil Materials and Clay	22
C-5	General Classification and Principal Mineral Assemblages of Igneous Rocks	35
C-6	Geologic Information Categories for Selection of a Site for High Level Radioactive Waste Disposal	58
C-7	Geosphere Migration Parameters Affecting Transport	94
C-8	I-129 Arrival Times and Relative Concentrations	98
C-9	Radionuclide Amounts for ORIGEN Output	99
C-10	Parameter Values Used in Scenarios 1 and 2	105
C-11	Scenario 2--Worst-Case Flow Actinides and Daughters After 300 Years	107
C-12	Scenario 2--Fission Products and C-14	112
C-13	Peak Values of I-129	114
C-14	Scenario 1--Diffusion Pathway - Actinides and Daughters After 6000 Years	117
C-15	Scenario 1--Diffusion Pathway - Fission Products and C-14	119
C-16	Actinide Solubility Limits	121

LIST OF TABLES (cont.)

<u>Table No.</u>		<u>Page</u>
C-17	Physical Data and Dose Commitment Factors for Significant Radionuclides	139
C-18	Radionuclide Concentrations Following Diffusion into a Potable Aquifer (Scenario 1)	140
C-19	Radionuclide Concentrations in Aquifer at Aquifer Entry (Scenario 2)	144
C-20	Environmental and Biological Transfer Coefficients for Dose Commitment Calculations	146
C-21	Information for River Model "A" for Reasonable Population Dose Calculations	148
C-22	Information for River Model "B" for Reasonable Population Dose Calculations	149
C-23	Fifty-Year Dose Commitment Calculated for the Case of Diffusion into a Potable Aquifer (Scenario 1)	159
C-24	Dose Commitment Calculated for Aqueous Transport by Flow into a Potable Aquifer (Scenario 2)	160
C-25	Summary of Total "Maximum" Individual and Population Dose Commitments for Direct Ingestion of Contaminated Aquifer	161
C-26	Summary of Reasonable Population Dose Commitments by Significant Radionuclides and Pathways	165
C-27	Comparison of "Reasonable" Population 50-Year Dose Commitments for Ra-226 from Scenario 2 and Ra-226 in the Colorado River	172

LIST OF FIGURES

<u>Figure No.</u>		<u>Page</u>
C-1	Ranges of Distribution Coefficients for Various Rock Types	21
C-2	Radioactive Waste Repository Conceptual Model in Tectonized Shale	41
C-3	Radioactive Waste Repository Conceptual Model in Salt Dome	44
C-4	Regions of the World in Which Major Earthquakes and Volcanoes Occur	52
C-5	Radioactive Waste Repository Conceptual Model in Bedded Salt	62
C-6	Radioactive Waste Repository Conceptual Model in Non-Tectonized Shale	67
C-7	Radioactive Waste Repository Conceptual Model in Basalt	70
C-8	Two Flow Regimes	73
C-9	Cross-Section Data for Evaluation of Flow Regimes	74
C-10	Calculated Interbed Velocities for Two Flow Regimes	76
C-11	Comparison of Laboratory and In-Situ Leach Data	81
C-12	Comparison of Leach Rates for Two Leach Models	84
C-13	Numerical Representation of Geologic Models	91
C-14	Effect of Leach Time and Migration on Nuclide Concentration	101
C-15	Schematic Illustration of Maximum Individual and Upper Limit Population Dose by Diffusion to a Potable Aquifer (Scenario 1)	141

LIST OF FIGURES (cont.)

<u>Figure No.</u>		<u>Page</u>
C-16	Schematic Illustration of Maximum Individual and Upper Limit Population Dose by Aqueous Flow Transport to a Potable Aquifer (Scenario 2)	143
C-17	Schematic Illustration of "Reasonable" Population Dose (Scenarios 1 & 2)	155

C-1.0 INTRODUCTION

In providing technical support of standards for high-level radioactive waste management, this Task C report has three principal objectives:

- To assess geologic site selection factors.
- To review available information and quantify the potential for the migration of nuclides through the geosphere to the biosphere, and finally
- To consider dose-to-man implications of a repository for high-level waste (HLW) containing large quantities of radionuclides in high concentrations that might become dispersed into the biosphere over geologic times.

The Task C work implicitly assumes that no repository is likely to meet all criteria that would ensure absolute integrity of the repository over geologic time or even over one or two centuries. Rather, repository integrity will involve:

- Selection of a site whose characteristics minimize, but do not entirely remove, the probability of nuclide migration,
- Site/repository engineering solutions to problems that cannot be avoided in the site selection process, and
- Site monitoring to allow remedial action should events require it.

Thus, Task C attempts to summarize the influences on nuclide migration potential and thereby identify critical inadequacies in the data and analytical method.

C-2.0 SUMMARY AND CONCLUSIONS

C-2.1 ASSESSMENT OF GEOLOGIC SITE SELECTION FACTORS

The extent to which geologic media can serve as barriers to radionuclide migration depends principally on the geologic properties of the media and on the possibility of catastrophic geologic events. Such factors as lithology, stratigraphy, structure, hydrogeology, tectonics, erosion, and mineral resources--and the interrelationships among these factors--must be identified and evaluated from the perspective of radionuclide transport or containment.

The "ideal" land site for a high-level radioactive waste repository would be geologically stable and isolated from aquifers and from the biosphere. The rock would be capable of adsorbing radionuclides and diffusing heat. Since an "ideal" site will not exist in nature, this task report identifies geologic factors that bear importantly on siting decisions in general.

Siting factors rely on a detailed knowledge of hydrogeologic, geochemical, thermal, and engineering properties of the repository material and of the surrounding rock. Of principal importance are the basic physical rock properties, e.g., permeability, porosity, and fracture systems that determine the basic hydrologic parameters of a rock suite, flow rate through the rock most particularly. Other rock properties, e.g., geochemical and thermal, operate to modify (largely to retard) the rate of hydrologic transport of various species, e.g., chemical constituents as typified by radionuclides. Engineering properties of the repository host rock taken as a whole, are critical to the repository itself and are, therefore, the most vital factor in the sense that they are the initial determinant of repository integrity.

In examining a specific site, a number of basic studies are required, each designed to yield information sufficient to identify and evaluate

site seismicity characteristics, rock type, and distribution; the structure or physical arrangements of the rocks; tectonic processes, e.g., volcanism; and surficial processes such as erosion. Inventories of mineral resources and other near-site activities such as oil and gas drilling could also bear on site suitability.

The notion of site factor ranking should be avoided in other than site-specific cases since, at any given site, one or another factor may predominate. To suggest a generic ranking assumes a uniformity and simplicity in nature that cannot be supported. For example, in a salt dome, the plastic behavior of salt under thermal stress over long periods of time is of greatest importance. In layered or bedded salt, the distribution and thickness of interlamination of clays and shales with the salt, as well as joint or fault systems, are of primary significance. In basalt terrains, interlayers of rubble that may be aquifers are important, as well as joint or fault systems that could serve as conduits from a repository to the biosphere. These several examples do not suggest that other factors can be neglected or in fact that in any given instance another factor or several factors might not predominate. These factors reflect the inherently complex character of crustal rock, which makes investigation difficult. This complexity accounts as well for success, or lack thereof, in drilling oil wells, unpredictable rock fall behavior in mines, and the profound difficulties encountered in earthquake prediction.

In order to facilitate understanding of these processes, generalized models were developed that formed the conceptual background, utilized subsequently in the Task C Report for analysis of migration from the repository through geologic media to the biosphere and for evaluation of the potential dose to man.

C-2.2 ANALYSIS OF MIGRATION POTENTIAL

The objective of this sub-task was to select a suitable model for quantitative prediction of the migration of radionuclides from a breached HLW containment through the geologic environment and into communication with biosphere pathways to human consumption. The intention of the task was not to develop entirely new physical and analytical models, but rather to review the applicability of existing models, and identify and select an appropriate one for examination of parameters. The physical parameters of the model were to be compatible with the several characteristics of acceptable geologic media for siting of an HLW, and were to be definable as independent variables.

A review of available models identified one as being generally adequate. Sensitivity analyses were performed on the several parameters of this model, and appropriate ranges of values were found for each parameter.

The simplified model for migration of radionuclides from the HLW through the geosphere assumes ultimate failure of the primary containment system, and proceeds to estimate the concentrations of individual radionuclides that will escape from the repository by (a) leaching into groundwater, (b) transport in and/or through the groundwater in the vicinity of the repository upward to an overlying aquifer, and (c) lateral transport in the aquifer to points at various distances from the repository site. Mechanisms considered to affect transport between the repository and the overlying aquifer include permeability of the rock, convective water flow through fractured rock and, alternatively, diffusion of dissolved nuclides through non-flowing water in that region. Mechanisms affecting transport in the aquifer include convective flow and retardation of ions by selective adsorption.

Factors considered to affect the concentrations of individual radionuclides occurring at various locations in the aquifer and at various points in time (up to 100,000 years after emplacement) include the mass of each nuclide originally present in the HLW, radioactive decay and growth, dilution by groundwater, and selective adsorption within the aquifer. Calculations were performed, leading to predicted concentrations

of specific radionuclides at various points in the aquifer up to 8.0 km (5 miles) distant from the repository. Only a few of the actinide and fission product nuclides were predicted to exist at significant concentrations at distances beyond 1.6 km (1 mile) from the repository at any time up to 100,000 years after emplacement. These nuclides were used as the basis for calculating dose-to-man effects. In order to compare the concentrations of nuclides predicted to migrate to various points at distances away from the repository with those that might be expected to be leached away from a naturally-occurring uranium ore deposit of comparable size and in the same location, calculations were made using the same assumptions of factors affecting migration, and assuming a 0.2% uranium ore deposit at the site of the HLW. The circumstances assumed for the HLW case (Scenario 2) resulted in approximately equal concentrations of Ra-226 in the aquifer as for the uranium ore case.

C-2.3 DOSE-TO-MAN CONSIDERATIONS

Considerations of alternatives for geologic emplacement of high-level waste (HLW) lead to a limited range of possible circumstances for transport of radionuclides away from the repository site, once the primary containment system has been breached. Two sets of conditions were selected for modeling the circumstances for escape of radionuclides to the biosphere. Those sets of conditions, labeled "Scenario 1" and "Scenario 2," were used to predict the concentrations of specific radionuclides occurring in potable aquifers at various distances from the repository site and at various times in the future.

Given the predicted levels of various radionuclides in the potable aquifer under the conditions assumed for "Scenario 1" (diffusive transport to the aquifer) and "Scenario 2" (direct connective flow transport to the aquifer), the possible rates for exposure of people to the radiation dose resulting from ingestion of contaminated food or of the water itself were examined. Calculations of anticipated human radiation dose commitments were made for three different hypothetical cases:

- Human ingestion, for a period of one year, of water taken from the contaminated aquifer at a point directly over the breached repository.

- Human ingestion of vegetables, milk from cows, and beef from cattle that have served as biological transfer agents of radionuclides from contaminated water, as well as some direct ingestion of the water itself. Use of water from the river, in this case, is predominantly for agricultural purposes, so radiation dose commitments to the affected population are dominated by crop-ingestion mechanisms. The contaminated water in this case is assumed to be from a hypothetical river (River Model "A") that intersects the contaminated aquifer at a point 1.6 km (1 mile) from the breached repository.
- Conditions identical to the above, except that more of the water from the hypothetical river (River Model "B") is used for direct human consumption than is the water from River Model "A."

Radionuclide concentrations in all three cases are calculated on the basis of each of the two assumed mechanisms for aquifer contamination, corresponding to Scenario 1 and Scenario 2, respectively.

The results of the analysis are given in terms of estimated 50-year radiation dose commitments to individuals and to populations of 250 people for the "maximum" dose assumption and 10,000 people for the "reasonable" dose assumption, respectively. Ingestion of the radionuclides is assumed to be for a period of one year, and dose commitments are calculated for whole body, for bone, and for thyroid.

The principal radiation dose effects were found to be from a small number of radionuclides, namely, strontium-90, iodine-129, radium-226, neptunium-237 and plutonium-239 and -240. Several other nuclides were included in the exercise, based on their apparent importance in terms of concentrations predicted in the aquifer at various times and places. When ingestion and dose commitment factors were included, however, only the named nuclides were found to contribute significantly to the dose-to-man estimates. The relative importance of each of these nuclides was found to change with time after emplacement. Comparisons were made of the predicted radiation dose effects of nuclides escaping from the HLW

with those of the natural background radiation. Except for the extraordinary circumstances postulated for calculations of the "maximum" individual dose commitment for Scenario 2, none of the modeled cases led to predicted dose commitments as large as that due to the natural background.

Comparisons were also made between the dose commitments predicted from the case of River Model "A" and those calculated from actual concentrations of radium in the Colorado River. (Characteristics of River Model "A" were based on actual characteristics of the Colorado River mainstream.) The radiation dose commitments estimated from present actual radium concentrations were found to be 10,000 times greater than those predicted for the maximum Ra-226 concentrations expected from the HLW leakage.

C-3.0 ASSESSMENT OF GEOLOGIC SITE SELECTION FACTORS

Task C-1 assesses deep geologic site selection factors and provides conceptual geologic models for conditions where HLW repositories may be located. The scope includes evaluation of the existing data on site selections, with no new development work. Much of the information is based on extensive studies conducted by contractors for the Energy Research and Development Administration, especially Battelle Northwest Laboratories and the U S. Geological Survey.

The following are included in this subtask:

- Evaluation of available pertinent published reports from the United States and abroad and discussions with recognized authorities in various applicable technical areas.
- Determination and discussion of intrinsic properties of geologic media which can serve as "barriers" and/or "paths" to radionuclide migration.
- Identification of geologic site selection factors, including lithology, stratigraphy, structure, hydrogeology, tectonics, erosion, and mineral resources.
- Development of an investigative sequence for evaluating site selection factors, considering their direct impact on radionuclide transport and interrelationships within the total geologic setting.
- Identification of general, but representative, geologic settings for developing illustrative conceptual geologic models of likely disposal areas, including bedded salt, salt domes, two types of shale, and a profile consisting primarily of basalt.

The major objective of disposal of HLW in deep geologic media is to provide containment until the radionuclides decay to an innocuous level. For this study, the required containment period is considered to be on the order of one million years. Containment is considered to mean

that no wastes (radionuclides) escape to the biosphere in concentrations deleterious to man or his environment.

The ideal site is one with an assured tectonic, hydrologic, and mechanical stability, which is isolated from any type of pathway-to-groundwater aquifers (rock or soil strata containing water that reaches the ground surface in any natural or manmade manner) and from the biosphere. The geologic materials should be capable of (1) adsorbing any radionuclides that begin to travel from their emplaced position and (2) diffusing heat without impairing the integrity of the natural formation.⁽¹⁾ Also, the ideal site would preserve for the longest period the integrity of any other containment methods used; namely, the vitrification of waste and the encapsulation of the solidified radionuclides into metal containers, as discussed in the Task B report. Individual sites that satisfy all individual objectives may not exist in nature, but a perfect site is not required so long as it has a proper combination of the related factors. This subtask identifies those geologic factors that will determine a site's suitability and shows how the factors can be balanced to satisfy the basic disposal objective. Non-geologic factors, including thermal and engineering properties, are discussed, but to a lesser extent.

Control of radionuclide release from a repository must account for several ground escape mechanisms:

- Catastrophic but nongeologic events, such as a meteorite impact, that could disrupt the repository.
- Catastrophic, geologic-related events, such as major fault displacement or volcanic eruption.
- Gradual processes which occur over long time periods.

This geologic site selection study covers only the latter two types of occurrences. Gradual release mechanisms include transportation of radionuclides through the geosphere. The transport mechanism can conceivably be through moving or stationary groundwater, moving or stationary gas within the pore spaces of the geologic media, or by diffusion

through the solid material itself. Available studies identify groundwater as the most important mechanism. Consequently, dispersion with moving groundwater and/or diffusion through stationary pore water are considered as the only significant mechanisms. These phenomena are largely controlled by the hydrogeologic and geochemical properties of the media as introduced in Section C-3.1.

Catastrophic geologic events are not dependent upon the intrinsic properties of the host rock at the repository. Discussions of these factors are included in Sections C-3.2 and C-3.3, where the site selection factors are introduced and evaluated.

Section C-3.4 identifies conceptually several types of settings for deep geologic repositories. It should be recognized that no unique site factor ranking scheme is preferable to another since the site factors are not analytically related in a mathematical or absolute sense. The basic utility of arraying factors is to assure completeness of the investigation.

C-3.1 IMPORTANT INTRINSIC PROPERTIES OF GEOLOGIC MEDIA

Unless a catastrophic event occurs, the escape of radionuclides will be dependent on such forces as the pressure, temperature, and chemical conditions surrounding the repository as well as on the basic intrinsic properties of the geologic media. Depending upon site-specific factors, one or several of the following intrinsic properties may take on greater importance.

The intrinsic properties of geologic media considered to be most important in relation to radioactive waste storage are:

- Hydrogeologic properties, which determine the forms in which water, water vapor, or gases exist in pore spaces and the rate(s) at which they will flow through a geologic medium.

- Geochemical properties, which play a major role in determining (1) the rate at which radionuclide movement is retarded or accelerated, (2) how the geologic material will behave when internal environmental conditions are changed, and (3) the nature of chemical reactions that may result from contact of radionuclides with the geologic media.
- Thermal properties, which determine the physical and chemical behavior of the media when heated and the rate at which heat can be dissipated.
- Engineering properties, which determine local structural behavior at the disposal opening before and after it is backfilled and, more importantly, how the media will respond to displacement changes from larger-scale geologic events.

The following sections further describe these intrinsic properties and how they influence radionuclide migration potential. Table C-1 presents typical ranges of certain intrinsic properties of several rock types.

C-3.1.1 Hydrogeologic Properties

Two basic hydrogeologic conditions govern the existence and/or movement of groundwater:

- The hydrologic properties and behavior of intact, unfractured or unbroken rocks (primary characteristics); and
- The hydrologic properties and behavior of disturbed zones (fractures, faults, joints, stylolites, solution channels, etc.) where the extent of disturbance, and not the rock itself, predominates (secondary characteristics).

Groundwater moves from levels of higher pressure to lower pressure. The velocity of the flow (for laminar flow) is proportional to the pressure drop per unit length of distance traveled, i.e., the hydraulic gradient. This relationship between the hydraulic gradient and the groundwater velocity is expressed by a relationship called Darcy's Law.⁽²⁾

TABLE C-1

EXAMPLE INTRINSIC PROPERTIES FOR TYPICAL REPOSITORY AND ASSOCIATED LITHOLOGIES

PARAMETER	SALT	LIMESTONE	SHALE	SANDSTONE	BASALT	GRANITE
Porosity (percent)	<1	15	10	<5-30	1-40	0.05-3
Permeability (cm/sec)	10^{-7} - 10^{-9}	10^{-5} - 10^{-9}	10^{-5} - 10^{-9}	10^{-2} - 10^{-7}	10^{-5} - 10^{-9}	10^{-10}
Thermal Conductivity (millilcal/cm sec°C)	13-17	5-8	3-7	3-10	3-4	6-9

Source: Ekren, E.B. et al., Geologic and Hydrologic Considerations for Various Concepts of High-Level Radioactive Waste Disposal in Conterminous United States. Open-File Report 74-158, U.S. Department of Interior Geological Survey, Richland, Washington, 1974.

Darcy's Law for a one-dimensional case is described by the equation:

$$V = K \frac{\Delta h}{L} \quad (1)$$

where

V = velocity of flow (cm/s),

K = permeability of the medium and indicates the ability of the fluid to move through the medium under a hydraulic gradient (cm/s),

Δh = difference in hydraulic head (cm),

L = the distance between the two points from which the hydraulic head change is measured (cm).

The ratio of the difference between hydraulic head (Δh) and the distance L is called the hydraulic gradient (I). To express the flow rate (Q), or the quantity of water flowing through a unit cross-sectional area (A) per unit time, Darcy's Law can be written:

$$Q = AV = KIA \quad (2)$$

The restriction that Darcy's Law is valid only under conditions of laminar flow and that it requires modification when turbulent flow prevails should not have impact on analyses for a suitable repository. Even though turbulent flow conditions may exist on a microscopic basis as groundwater moves through geologic media, the macroscopic outcome of flow in overall low permeability regimes (such as should exist in the geologic media surrounding any seriously-considered site for a repository) may be adequately described by the laws governing laminar flow (Darcy's Law).

The permeability used in the above equation may be (1) the primary permeability of the porous rock medium; or (2) the secondary permeability of the rock if flow is predominantly through fractures, fissures, or other types of small discontinuities.

The magnitude of the primary permeability is most dependent upon the grain size, the degree of crystallization, and/or cementation and compaction. The lower end of the permeability ranges shown for various rock types in Table C-2 is primary permeability.

Secondary permeability (resulting from cracks and fissures in the rock) is likely to be of greater concern when considering potential groundwater flow through strata surrounding a HLW repository. Secondary permeability must usually be determined in the field from well-planned boring drilled to intersect the predominant fractures in the rock. Careful attention must be given to the fracture orientations to facilitate recognition of preferred directions of permeability. Established relationships exist for deducing overall permeability values from field permeability tests and pump tests. Also, if appropriate measurements can be obtained from rock corings, the following equation can be used to estimate the secondary permeability value in fractured or fissured rock:⁽³⁾

$$K' = \frac{ge^2}{12\nu b} \quad (3)$$

where

K' = equivalent permeability of a planar array of parallel smooth cracks (cm/s),

g = gravitational acceleration (981 cm/s²),

e = opening of cracks or fissures (cm),

ν = the coefficient of kinematic viscosity
(0.0101 cm²/s for pure water at 20 °C.,)

b = spacing between cracks (cm).

For calculating flow through fractured or fissured rock, the permeability in Equation (1) should be replaced by K' (secondary permeability).

The flow velocity defined by Equation (1) is actually only an apparent velocity across a given cross section, considering that the water flows in a straight path. Actually however, the water flows in a very tortuous path around individual grains or fissures and into and out of pore and opening sizes of various sizes and configurations. Accordingly, in order to describe properly phenomena that can retard the migration of chemical elements through geologic media (discussed in Section C-3.1.2), it is necessary to introduce another term, interstitial velocity, that describes groundwater flow. As an example for porous media, interstitial velocity is defined by the equation:

$$V_w = V/\epsilon \quad (4)$$

where

V_w = interstitial velocity of groundwater flow
(cm/s),

ϵ = porosity (the ratio of the volume occupied by pores to the total volume of a geologic material).

Interstitial velocity is always greater than the Darcy velocity (Equation 1) when considering a specific hydrogeologic flow regime.

C-3.1.2 Geochemical Properties

Geochemical properties of rock at a repository are important for two reasons:

- Properties such as ion-exchange capacity, mineralogy, and natural water (pore water) chemistry can remove nuclides (sorption) from slowly moving water, reducing the threat to man even if absolute impermeability cannot be confirmed.
- Rock mineralogy, solubility, water content, and composition are important considerations in assessing potential waste-rock interactions--considering both the impact on the waste and on the rock.

C-3.1.2.1 Sorption Capacities of Geologic Materials

Section C-3.1.1 indicates that if water and a hydraulic gradient exist, some small flow can occur, even in very tight rock formations. This section shows that geologic media can act to impede radionuclide migration to the biosphere, even if flow does occur. Dissolved radionuclides may enter into complex physicochemical reactions with the geologic materials by phenomena such as adsorption, ion exchange, colloid filtration, reversible precipitation, and irreversible mineralization. These mechanisms are usually combined into one general term called "sorption." In effect, the sorption phenomena cause ions to move at much lower velocities than the medium (groundwater in this case) transporting them.

Sorption is expressed in terms of distribution coefficients, K_d for porous media and K_a for fractured media, as follows:⁽⁴⁾

$$K_d = \frac{f_m \times L_v}{f_s \times M_w} \quad (5)$$

where

K_d = distribution coefficient for porous media in
ml/g,

f_m = the fraction of total activity adsorbed on the mineral,

L_v = volume of solution equilibrated with M_w (ml),

f_s = fraction of total activity in solution = $1 - f_m$,

M_w = weight of mineral (g).

and

$$K_a = \frac{f_m \times L_v}{f_s \times F_a} \quad (6)$$

where

K_a = distribution coefficient for fractured or faulted
media in ml/cm²

F_a = surface area of both sides of the fracture
(cm²).

The K_d and K_a values are dependent in a complex way on such parameters as pH of the groundwater, the specific nuclides present, concentration and type of dissolved ions, temperature, oxidation-reduction potential (redox potential, E_H), presence or absence of organic material, and others.

The effectiveness of any geologic medium to act as a retarder for a particular condition is expressed as the retardation factor, R_d . For a particular element, R_d is defined as the ratio of the water velocity to the nuclide migration velocity (dimensionless term). The retardation factor is related to the distribution coefficients by the following relationships:

$$\text{For porous media, } R_d = 1 + K_d \rho / \epsilon \text{ (dimensionless)} \quad (7)$$

$$\text{For fractured media, } R_d = 1 + K_a R_f \text{ (dimensionless)} \quad (8)$$

where

ρ = bulk density of the medium (g/cm^3),

ϵ = porosity as defined in Section C-3.1.1 (dimensionless),

R_f = the surface-to-volume ratio of the fracture (cm^2/ml).

The magnitude of radionuclide migration retardation that can be realized may be expressed by relating the velocity of ions moving through the geologic medium to the interstitial velocity of water flow by the following equation:

$$V_i = \frac{V_w}{R_d} \quad (9)$$

where

V_i = the velocity of the ionic species,

and all other terms are as previously defined.

As discussed below in relation to Table C-3, retardation factors can vary from as low as 1.0 to as high as 50,000 or more. For this range, the velocity of the ions can be the same as the water transporting them or as little as 10^{-5} times as fast.

TABLE C-3

ESTIMATED DISTRIBUTION COEFFICIENTS (K_d) AND
RETARDATION FACTORS (R_d) IN A TYPICAL DESERT SOIL

ATOMIC NO.	ELEMENT	K_d (ml/g)*	R_d †	V_i/V_w ‡
1	Tritium	0	1	1
4	Beryllium	75	330	3×10^{-3}
6	Carbon	2	10	1×10^{-1}
11	Sodium	10	50	2×10^{-2}
17	Chlorine	0	1	1
18	Argon	0	1	1
19	Potassium	35	170	6×10^{-3}
20	Calcium	15	100	1×10^{-2}
26	Iron	150	3,300	3×10^{-4}
27	Cobalt	75	330	3×10^{-3}
28	Nickel	80	330	3×10^{-3}
34	Selenium	20	100	1×10^{-2}
36	Krypton	0	1	1
37	Rubidium	125	500	2×10^{-3}
38	Strontium	20	100	1×10^{-2}
39	Yttrium	2,000	10,000	1×10^{-4}
40	Zirconium	2,000	10,000	1×10^{-4}
41	Niobium	2,000	10,000	1×10^{-4}
42	Molybdenum	5	25	4×10^{-2}
43	Technetium	0	1	1
46	Palladium	250	1,100	9×10^{-4}
48	Cadmium	2,000	10,000	1×10^{-4}
50	Tin	250	1,100	9×10^{-4}
51	Antimony	15	100	1×10^{-2}
53	Iodine	0	1	1
55	Cesium	200	1,000	1×10^{-3}
61	Promethium	600	500	4×10^{-4}
62	Samarium	600	2,500	4×10^{-4}
63	Europium	600	2,500	4×10^{-4}
67	Holmium	600	2,500	4×10^{-4}
81	Thallium	2	10	1×10^{-1}
82	Lead	4,000	16,700	6×10^{-5}
83	Bismuth	10	50	2×10^{-2}
84	Polonium	25	110	9×10^{-3}
85	Astatine	0	1	1
86	Radon	0	1	1
87	Francium	200	1,000	1×10^{-3}
88	Radium	100	500	2×10^{-3}
89	Actinium	1,000	5,000	2×10^{-4}
90	Thorium	15,000	50,000	2×10^{-5}
91	Protactinium	4,000	16,700	6×10^{-5}
92	Uranium	3,000	14,300	7×10^{-5}
93	Neptunium	15	100	1×10^{-2}
94	Plutonium	2,000	10,000	1×10^{-4}
95	Americium	2,000	10,000	1×10^{-4}
96	Curium	600	3,300	3×10^{-4}
97	Berkelium	700	3,300	3×10^{-4}

NOTES:

*Equilibrium distribution coefficients between water and soil.

†Retardation Factor (R_d) = V_w/V_i

‡Relative transport rate of elements in assumed soil to that in water.

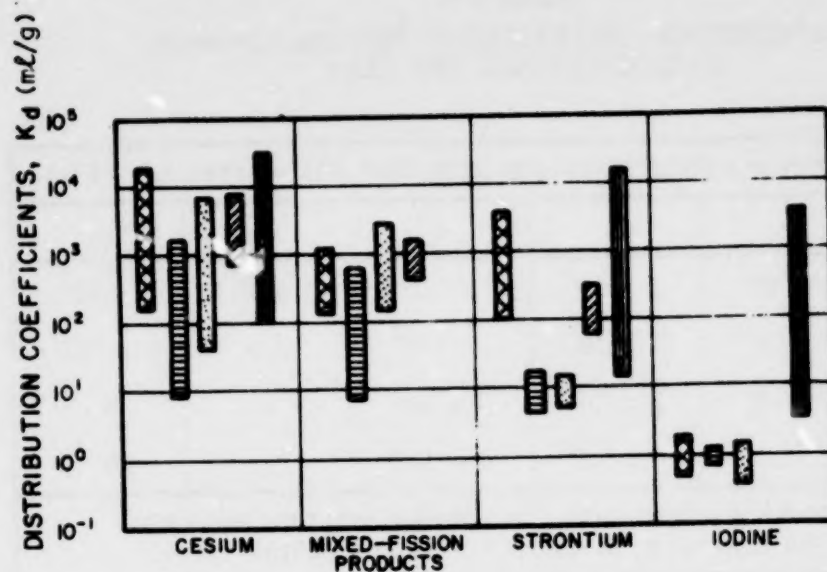
Source: Modified after Denham et al. (6).

In the mechanism that permits sorption to function, exchangeable ions are released from the host rock into the water in exchange for radionuclides. As an example, the adsorption of cesium or strontium ions by clay can release sodium or potassium ions into the water. Thereafter, those released ions, together with ions originally present in the groundwater, compete with radionuclides for additional available exchange sites.⁽⁵⁾ This process allows radionuclide migration to be greatly retarded for many conditions, but not necessarily eliminated.

Distribution coefficients are variable and uniquely determined for only a single set of physical and chemical conditions. Figure C-1 illustrates ranges of distribution coefficients for tuff, granite, limestone and dolomite, basalt, and soils in relation to cesium, mixed fission products, strontium, and iodine.⁽⁵⁾ For the cesium, mixed fission and strontium cations (positively charged ions), granite possesses the lowest distribution coefficient (10^1 to 10^3 ml/g), while tuff and soil exhibit the highest coefficients (10^1 to $10^{4.5}$ ml/g). It is noteworthy that for the iodine anion (negatively charged ion), the distribution coefficient is close to 1 for the rock types, but from about 10^1 to $10^{3.5}$ ml/g for soils. Although very little research has been conducted on anionic solutions, the available data indicate that the mechanisms controlling retardation of cations may not readily apply to retardation of anions.⁽⁵⁾

As further related examples, Table C-3 lists estimated distribution coefficients for a typical desert soil for a wide variety of nuclides.⁽⁶⁾ This table shows that the K_d can be as low as zero for tritium, various noble gases, iodine and a few other elements or as high as 4000 and 15,000 ml/g for protactinium and thorium, respectively.

Table C-4 indicates distribution coefficients for different soil materials and for clay samples taken at various depths for plutonium, americium, yttrium, strontium, and cesium.⁽⁷⁾ For these materials, strontium has a very low K_d , ranging from 2 ml/g in river sand to about



NOTE:

DISTRIBUTION COEFFICIENT VALUES
FOR IONS SHOWN

LEGEND

- TUFF
- GRANITE
- LIMESTONE AND DOLOMITE
- BASALT
- SOILS

SOURCE:

SCHNEIDER, K. J. AND A. M. PLATT (EDS.)
HIGH-LEVEL WASTE MANAGEMENT ALTERNATIVES.
BNWL-1900, BATTELLE PACIFIC NORTHWEST
LABORATORIES, RICHLAND, WASHINGTON, MAY 1974.

FIGURE C-1 RANGES OF DISTRIBUTION COEFFICIENTS
FOR VARIOUS ROCK TYPES

TABLE C-4
DISTRIBUTION COEFFICIENTS (K_d) FOR DIFFERENT
SOIL MATERIALS AND CLAY

DISTRIBUTION COEFFICIENTS FOR DIFFERENT SOIL MATERIALS (mL/g)					
MATERIAL	Pu	Am	Y	Sr	Cs
Clay	10^4	5×10^4	10^3	2.5-10	2×10^2
Sandstone	2×10^3	2×10^4	10^3	2.5	2×10^2
Caprock	5×10^2	3×10^3	6×10^2	4	6
River sand	2×10^2	4×10^2	6×10^2	2	10
DISTRIBUTION COEFFICIENTS FOR VARIOUS NUCLIDES MEASURED FOR CLAY SAMPLES TAKEN AT VARIOUS DEPTHS (mL/g)					
SAMPLE DEPTH (meters)	Pu	Am	Y	Sr	Cs
20 - 30 *	10^4	8×10^4	1.6×10^3	21	3×10^2
55 - 60 †	5×10^3	6×10^4	8×10^2	20	2×10^2
100 - 125 †	6×10^3	5×10^4	6×10^2	12	2×10^2
120 - 130 †	5×10^3	6×10^4	7×10^2	4	2.7×10^2
120 - 160 ‡	7×10^3	2×10^4	6×10^2	5	3×10^2
200 - 225 ‡	6×10^3	9×10^4	8×10^2	7	1×10^2
245 - 275	8×10^3	4×10^4	7×10^2	4	1.1×10^2
300 - 325	9×10^3	4×10^4	7×10^2	3	2×10^2
328 - 348	8×10^3	5×10^4	8×10^2	2.5	2.5×10^2

NOTES:

* Quaternary clay.

† Young Tertiary clay.

‡ Older Tertiary clay.

SOURCE: Modified from Hamstra, J. and B. Verkeek. Review of Dutch Geologic Waste Disposal Programmes. International Conference on Nuclear Power and its Fuel Cycle, Salzburg, Austria, 2-3 May 1977. IAEA. CN36/209, Netherlands Energy Research Foundation ECN Patten, the Netherlands, 1977.

20 ml/g in clay material. Plutonium and americium have the largest distribution coefficients, reaching their maximums in clay at around 10^3 to 10^4 ml/g. The distribution coefficient values due to sorption are higher for all of the radionuclides in clay material. No relationships between K_d and depth are apparent from the data in Table C-4.

None of the distribution coefficients presented above as examples can be used directly in determining the retardation factor in fractured or fissured media. Little work has been done to determine these values and representative values are not reported in the literature. However, the distribution coefficient for fractured media for a particular nuclide should be expected to be significantly lower than the distribution coefficient for porous media for the same ion based only on the geometry of the two types of flow.

Retardation factors are also presented in Table C-3 for the corresponding distribution coefficients. These are calculated for specific media porosities and bulk densities for illustrative purposes only. Considering the R_d for thorium (50,000) as an extreme example and using Equation (9), it is apparent that the velocity (V_1) at which thorium would move through the tested material is a factor of 2×10^{-5} lower than the interstitial velocity (V_w) of the groundwater transporting it. For this case, radon would move with the same velocity as the groundwater and cesium would move at only 10^{-3} times the rate of the groundwater.

The presence of organic material in soil has been found to reduce K_d values significantly because of complexing for some radionuclides. The influence of organic matter on radionuclide retention is an important consideration that needs considerable attention at this time. Recent studies with cobalt have shown that while cobalt has a relatively high K_d value in soil free of organics, the K_d value drops to nearly zero as a result of the cobalt complexing with organic material.

The oxidation potential (E_H) is important for many of the radionuclides. Depending upon oxidation state, some of the nuclides can exist in a cation or anion form. In anion form, or a complexed form such as mentioned above, the K_d value may be rather low. In the cation form, the same nuclide may be strongly retained by the rock or be limited by its solubility. As a consequence, it is extremely important in making K_d measurements to recognize the oxidation potential of the natural or modified groundwater environment. Examples of nuclides that could be affected in terms of retention and/or solubility include many of the actinides as well as technetium.

The above discussion shows that lithologic types possessing high "sorption" capabilities could have significant advantages for purposes of radionuclide retention in the geosphere, all other factors being equal. The sorption capabilities expected for fractured or fissured media where groundwater flow could be most important to containment may be significantly lower than the examples presented in Tables C-3 and C-4 for porous media, but this phenomenon would still be very important in the overall potential for geologic materials to act as barriers to radionuclide migration.

C-3.1.2.2 Potential High-Level Waste-Rock Interactions

The two general categories of interactions considered under this section are:

- Reactions caused by the geologic materials comprising the repository horizon or their associated pore fluids, which have a corrosive effect on the waste containers and/or the waste form.
- Reactions induced by heat and radioactivity generated by the waste material, which have a negative impact on the repository stratum.

As discussed in Task B of this study, HLW may be placed into a repository as a vitrified solid and/or in metal canisters. Both of these factors add containment capability beyond that provided by the geologic media. Accordingly, there is benefit in maintaining the integrity of the primary containment systems, particularly the vitrified state, as long as

possible, thereby effectively restricting the mobility of most of the nuclides. However, the host materials or their associated pore fluids may be naturally corrosive, and chemical reaction (corrosion) rates usually increase with temperature. A special discussion of this issue with respect to salt follows in Section C-3.1.2.3. Therefore, the longevity of both the waste containers and the waste form will be a function of the geochemical environment present in the repository. The significant geochemical parameters are those associated with chemistry, heat, and radioactivity.

Typically, high ionic strength solutions (solutions with large amounts of dissolved solids) have a higher corrosion potential (or dissolving ability) than solutions with lesser concentrations of dissolved solids.^(8,9,10) Consequently, groundwater high in dissolved solids may be expected to attack and dissolve both the HLW and its container more rapidly than would dilute groundwater. An example of a high-ionic-strength solution is sodium chloride brine, which can be associated with salt deposits and is not uncommon in many geologically old sandstone aquifers. An overall evaluation of site suitability should consider how the site conditions will affect the time required for the waste form to deteriorate and for the nuclides to become "free" to move if a transport medium or means exists.

The other major category of potential waste-rock interactions is concerned with the effect of waste-generated heat and radioactivity on the repository stratum. The most important effects concern the dehydration of hydrated mineral phases in surrounding rocks, the resultant water expulsion, and the potential for rock melting should insufficient heat dissipation occur.⁽⁵⁾

Reactions that occur by either of the above mechanisms at elevated temperatures are similar to igneous and metamorphic processes that occur in nature at great depths in certain zones of the earth's crust. As the temperature increases, originally stable minerals eventually recrystallize to a different group of minerals, which are stable under the new pressure/temperature conditions. In the presence of water, this recrystallization is generally rapid and extensive, and the water is often driven away from the reaction site (as a hydrothermal solution), carrying with it volatile components of the

reacting rock mass. In a repository, the expulsion of water (as a liquid or vapor) could provide a mechanism for transporting waste components as well as a wide variety of trace elements. For example, ions of elements with radioactive isotopes (as in HLW) are known to occur naturally in hydrothermal solutions that have passed through several hundred meters of nearly impermeable rock during geologic times and have eventually been dispersed to the environment through such mechanisms as groundwater mixing and hot springs.⁽⁵⁾ These natural phenomena indicate that the amount and characteristics of pore water and the hydrated mineral phases of the host rock must be considered in site evaluation.

For the unlikely situation in which rock materials adjacent to the repository actually melt from heat buildup, it may be expected that recrystallization will occur as the molten mass migrates a short distance away from the heat source. If radionuclides are incorporated into the melted rock material they would become part of the crystal structures of newly formed mineral phases, providing a relatively insoluble form for the radionuclides.

C-3.1.2.3 Salt Solubility

The geologic occurrence and solubility of salt can lead to geochemical consequences that could bear significantly on the integrity of an HLW repository. The potential problems seem more acute in the case of bedded salt as opposed to diapiric salt. The former is characterized by heterogeneity, with interbeds of thin shales, other evaporites, and limestone, the result of depositional processes. The geometry and precise geologic detail of the interbeds are difficult to predict or even to map over any significant distance.

The Ad Hoc Panel⁽¹¹⁾ noted, with respect to bedded salt that, "On a microscopic scale it can be seen that many halite crystals contain liquid (brine) or solid (anhydrite) inclusions. A recent report by Roedder and Belkin⁽¹²⁾ estimates the volume of these inclusions to be less than 1% but 'an additional possibly even greater volume % fluid is present in situ, filling intergranular pores'." The Panel noted the work of Bredehoft⁽¹³⁾ et al., which showed that... "at 200°C, water in rock salt including potassium and magnesium components can contain approximately

70 wt% of dissolved salts. This implies that as little as 1 wt % H_2O in a salt bed may yield fluid that is 3 wt % because of the high concentration of the solution. The dissolving power of such brines should not be underestimated." The Panel directs attention to... "the importance of knowing the water content of salt beds proposed for repositories, particularly with the background and experience at Lyons, Kansas, where considerable volumes of water migrated in an unpredicted manner."

Salt domes of the piercement variety are often referred to as ideal for an HLW repository. The salt is generally thick, though laterally limited, and the lithology tends to be uniform and typically free of open fractures. Most, though not all, domes appear to be remarkably dry and to have been so for millions of years. Technology is available for discriminating between "dry" and "wet" domes.

C-3.1.3 Thermal Properties

Considerable heat generated by the HLW materials will increase the ambient temperature at the depth of the disposal horizon. Therefore, the geologic formation should possess thermal properties that (1) promote heat dissipation, (2) do not have mineralogic and pore water characteristics that will create unacceptable behavior changes or water migration during heating, or (3) do not become unstable because of heat buildup prior to the time that backfilling occurs.

The thermal conductivity of the rock is a primary consideration. A relatively high thermal conductivity is desirable to dissipate the heat produced by radioactive decay efficiently and steadily with disposal canisters located at a reasonable spacing. Typical thermal conductivities of potential host geologic rocks are shown in Table C-1.⁽¹⁴⁾ These values indicate that salt has the highest thermal conductivity of the rock types listed, followed by quartzite, granite, shale, and tuff in decreasing order.

Salt will also exhibit higher rates of creep at elevated temperatures than other rock types. This creep can create an increased engineering problem during disposal operations. On the other hand, it may be very beneficial because it promotes sealing and healing of fractures around the disposal cavity and of backfilled salt rock. Accelerated deforma-

tion at elevated temperatures of more brittle rock could be detrimental because it might lead to increased fracturing.

C-3.1.4 Engineering Properties

The basic engineering properties of rock, such as strength and stiffness, have obvious importance to the basic "mining engineering" requirement to establish repository openings and maintain their integrity until backfilling is accomplished. This factor is very important from an economic viewpoint, but does not have significant impact on the ability to assure containment.

Elastic-behaving rocks, like salt and some shales, will provide some additional containment attributes as opposed to brittle-type rocks, if movement of the repository formation does occur due to any external event. The more flexible rocks can deform to a greater degree without cracking, whereas brittle rocks usually fracture under any significant geologic movements. Fracturing to any extent can serve to reduce the containment capability of the repository horizon.

C-3.2 IDENTIFICATION AND DISCUSSION OF SITE SELECTION FACTORS

The perspective on geologic site selection criteria depends, in the absence of specific guidelines, on the viewpoint of the investigator(s). For example

- The designer responsible for assuring safe working conditions in the repository may be most interested in rock strength.
- The ventilation engineer may be most interested in pore water and its liberation into a high temperature environment--but only until each area is backfilled.
- The health physicist will be most interested in the overall containment capabilities of the total site for a very long period.

Clearly, the design team must carefully evaluate each site situation so that all required actions to ensure site integrity are provided for, including comparative evaluation(s) with alternative sites.

The natural complexity of geologic environments and the interrelated and interdependent nature of rock features will make each potential site unique and require that each site be evaluated individually. Con-

sequently, exact values, or even ranges of values, for site selection criteria cannot be specified in a generic study such as this. The purposes of the following discussions and Section C-3.3 are to identify site selection criteria applicable to most sites and establish a rationale for their relative ranking on a site-by-site basis.

The "design life" for containment for at least several hundreds of thousands of years must be emphasized throughout the site selection process. Obviously, when present-day site evaluations are conducted, the ability to predict the timing and magnitude of particular geologic occurrences diminishes for extreme time periods. On the other hand, however, even a million-year period is considered relatively short in most studies of the geologic past.

The newness of investigations for underground disposal of HLW and their greater-than-normal dependency on the potential occurrence of geologic events permit the listing of site selection factors to be arranged in several logical ways. The following arrangement seems logical for this study:

Site Selection Factors

Geologic Properties:

- Lithology (Rock Types)
- Stratigraphy
 - Thickness and Lateral Extent of Strata
 - Depth of Repository Horizon
 - Uniformity and Homogeneity of Geologic Media
 - Nature of Overlying, Underlying and Flanking Strata
- Structural Geology
 - Folding
 - Joints and Faults
 - Major Plastic Flows
- Hydrogeology
 - Regional Hydrogeology
 - Local Hydrogeology

Geologic Events:

- Tectonics
 - Tectonism
 - Seismicity
 - Diapirism
 - Volcanism

- Erosion and Denudation

- Water Flow
 - Glaciation

- Mineral Resources

Each of these items is related to time considerations and to intrinsic rock properties. Two additional descriptive categories--geometry and geography--were also considered as major elements in the selection factor list. For this study, however, they are both incorporated into more traditional geologic terms. A different list, incorporating these and other logical breakdowns would not conflict with the following discussions. Mineral resource considerations are obviously not a geologic event. They are listed in the event category, however, because in the context of an HLW repository, mineral resources represent a condition that could lead to an event with some impact. Each property, event, and term is defined and/or discussed in the following subsections.

Establishing the site values for each of the above factors would require very extensive and probably site-descriptive studies including seismic surveys, core drillings, and others. A question arises with respect to the potential for rendering the site reusable by the very studies designed to illustrate its integrity. Clearly, a balance must be struck reflecting this conflict, which implies that a residual but acceptable level of uncertainty (e.g., risk) will remain.

C-3.2.1 Lithology

Lithology is a very important site selection factor, in that it encompasses the detailed description of the rock's basic characteristics, including mineralogic composition, grain size and cementation, and other properties that determine how the material will behave when exposed to external forces or changes in environmental conditions. Rock types that are representative of lithologies being considered for HLW repositories include salt, shale (argillaceous materials), limestone, and igneous rocks. The following paragraphs summarize the salient features for each of these lithologic types or groups. Individual characteristics of any particular site will often vary from these capsule discussions, but the basic features discussed represent a meaningful starting point.

C-3.2.1.1 Salt

Salt (NaCl or halite) is an evaporite type rock, originally formed in basins that became isolated or partially isolated from the sea. The isolation allowed evaporative concentration and then salt precipitation to occur. Salt is often found as relatively pure beds, but thin layers are sometimes interbedded with other rock types or other evaporites.

Stratigraphic and structural features of salt formations are discussed further in the next two sections. It is notable here that both bedded salt and dome salt (formed from diapirism of deep bedded salt) are being considered for HLW disposal.

Several basic properties of salt that are attractive for the location of a repository are:

- Low porosity,
- Low permeability,
- Plastic behavior for healing fractures, and
- High thermal conductivity.

Some other characteristics of salt can cause special concern:

- High solubility,
- Mineral impurities in some deposits,
- High corrosion potential, and
- Low ion-exchange capacity.

Another unique feature of salt is the potential presence of brine cavities filled with salt-saturated water. It has been shown that upon heating, the brine cavities can migrate toward the source of heat.⁽¹⁴⁾ The mechanism for this migration is that the material at the hot side of the cavity (the side nearest the heat source) goes into solution, while salt precipitates at the cold side. In some locations, salt brine migration may be the only possible source for the natural introduction of water into a repository.

Another potential natural source of water in many salt deposits may be from hydrous mineral phases present as impurities in the formation.

Water present in many of these minerals such as polyhalite and gypsum can be released at only moderate temperatures. Gypsum, for example, will yield approximately 75% of its water when heated to 190 or 200°C. Water represents about 21% of the gypsum; if this mineral is present in a large quantity, it could release a significant volume of water into the repository. However, phase behavior of minerals under repository pressure and temperature conditions will not be the same as for ambient conditions. Therefore, water release based on assumptions of ambient conditions may, or may not, model conditions in or near the repository.

C-3.2.1.2 Shale

Shale is a general term for lithified clay or mud, although most shale does contain varying amounts of silt and sand material. Some shales have several properties that make them attractive for an HLW repository, including:

- Low permeability,
- Relatively good plastic behavior (though not as good as salt),
- High ion-exchange capacity, and
- Wide geographic distribution of nearly homogeneous material.

Potential negative characteristics of shale that must be investigated include:

- Many argillaceous rocks, especially those with relatively good plastic behavior, contain large quantities of hydrous minerals, such as montmorillonite that will release water when heated.
- Some shales contain petroleum or large amounts of carbonaceous material that may yield combustible carbon compounds when heated.
- Many shales are interbedded with more permeable rock types such as sandstone, limestone, and coal.
- Most shales possess relatively low thermal conductivity.

From the important viewpoint of water flow, marine shale sequences are generally free of interbedded coarser-grained rocks and can be less

permeable than continental (interbedded) sequences. Shales are not self-healing like salts, however, and investigations to evaluate the potential for groundwater flow must consider the potential for faulting or fracturing in the future, as well as the impact of fractures, joints or faults on secondary permeability (Section C-3.1.1).

C-3.2.1.3 Limestone

Limestone is a carbonate rock composed primarily of the mineral calcite (CaCO_3), usually with varying amounts of the mineral dolomite, $\text{CaMg}(\text{CO}_3)_2$. When the latter is dominant, the rock is generally referred to as dolomite. Limestone can be attractive because:

- It sometimes exists over large areas as a thick homogeneous stratum,
- Its basic matrix is fine grained, so the rock can have a very low primary permeability.

Several characteristics of limestone that may not be favorable are:

- Carbonate rocks typically exhibit greater solubility than other non-evaporite type sedimentary rocks. Solutioning can greatly increase secondary permeability and, in turn, the overall capacity of a limestone formation to transmit groundwater.
- Limestone is usually strong but brittle, so fracturing may occur from regional tectonic movements.
- Limestone does not possess significant sorptive properties except in areas of solutioning where clay minerals are commonly concentrated. Since these areas of solutioning also represent avenues for groundwater flow, the sorptive capacity provided by clay filling may partly counterbalance the slow groundwater movement through the solution channels.

A somewhat special consideration associated with disposal of HLW in carbonate rock is the potential for release of carbon dioxide gas (CO_2) from the calcite and dolomite mineral phases. This could be

caused by excessive heat buildup and could ultimately result in the availability of gas as a transport mechanism.

C-3.2.1.4 Igneous Rocks

The term igneous rocks refers to a group of rocks formed by the solidification of molten rock. This group may be subdivided into intrusive-type (plutonic) rocks, referring to those rocks which have solidified below the earth's surface at various depths, and extrusive-type (volcanic) rocks, which solidified after eruption onto the earth's surface.

Intrusive rocks can range from fine-to coarse-grained, depending upon the depth of intrusion, rate of cooling, and presence of gaseous constituents; extrusive rocks are fine-grained or glassy and consist either of rocks formed from lavas that flowed onto the earth's surface or from volcanic ash that either flowed or was deposited on the earth's surface. Table C-5 shows the general classification and principal mineral assemblages of common plutonic and volcanic rocks.⁽⁵⁾ It also shows the approximate percentages of silica contained in each general type of rock.

Igneous rocks vary in hardness and strength, depending upon crystal size, the mode of eruption, and the presence or absence of voids. Plutonic rocks have generally low porosities and permeabilities, whereas the volcanic rocks vary in these properties from very low to extremely high. Plutonic rocks consist largely of granite, granodiorite, diorite, and gabbro. These rock types can generally be considered similar from the standpoint of mechanical strength and overall suitability for waste disposal. Granites, however, have the lowest melting points and are somewhat weaker than the other rocks in this category. Volcanic rocks may be chemically similar to the intrusive varieties but differ in their textures and physical properties.

Favorable characteristics of igneous rocks for the location of an HLW repository are:

TABLE C-5
GENERAL CLASSIFICATION AND PRINCIPAL
MINERAL ASSEMBLAGES OF IGNEOUS ROCKS

TEXTURE	>66 PERCENT SILICA (ACID)	52-66 PERCENT SILICA (INTERMEDIATE)	<52 PERCENT SILICA (BASIC)
Plutonic (medium- and coarse-grained).	Granite	Granodiorite	Gabbro
Volcanic (fine-grained or glassy lava or tuff).	Rhyolite	Quartz latite and rhyodacite	Basalt and basaltic andesite
	Major minerals-- quartz, potassium feldspar, mica.	Major minerals-- quartz, sodium- calcium, feldspar, mica, amphibole.	Major minerals-- calcium-sodium feldspar, pyroxene, olivine.

Source: BNWL-1900 (5)

- High mechanical strength,
- Low primary permeability,
- Low porosity,
- The occurrence of thick rock bodies (with the exception of certain volcanic flow deposits).

Unfavorable characteristics for the location of an HLW repository include:

- Potential for high secondary permeability due to jointing and faulting,
- Low sorption potential (except for tuffaceous deposits),
- Relatively low thermal conductivity for heat dissipation.

For purposes of this study, basalt has been chosen to represent a typical igneous rock type. Section C-3.4 will discuss a typical geologic setting in which basalt occurs in suitable geometry for location of a repository.

C-3.2.2 Stratigraphy

When considering any particular lithologic unit for location of an HLW repository, it is necessary to understand the host material itself and its relationship with surrounding geologic materials and conditions. The study of the overall spatial and temporal relationships of rock formations is called stratigraphy.

Determination of stratigraphy is accomplished by a wide variety of studies, including review of existing site and regional geologic information, drilling and field testing programs, and detailed data analysis. Important stratigraphic relationships are typically presented graphically as two-dimensional geologic cross sections or as block diagrams that provide a three-dimensional perspective. The pertinent characteristics of the conceptual cross sections prepared for this study are discussed in Section C-3.4.

Important stratigraphic elements relative to HLW repository sites are thickness and lateral extent of the host formation, depth of the disposal zone, uniformity and homogeneity of the formation, and the nature and extent of underlying, overlying, and flanking strata. These parameters are discussed in the following paragraphs.

C-3.2.2.1 Thickness and Lateral Extent of Strata

A host rock formation continuous for very long distances in both the horizontal and vertical direction would be ideal for almost all design criteria. This type of situation will probably not occur naturally, however, and the actual spatial extent of a host formation will be a major investigation and design factor. For example, current detailed siting investigations are considering both bedded salt and shale of moderate thickness and great lateral extent, and salt domes with pronounced lateral limits but thicknesses greater than 10,000 feet. The minimum acceptable thickness and lateral extent of any particular geologic formation will depend on such characteristics as the thermal properties of the formation. A critical criterion for formation size is that it be sufficiently large (thick and/or extensive) to assure adequate heat dissipation. For example, thinner deposits of salt may be utilized, whereas other rock types, such as basalt, may need to be considerably thicker because of their lower heat dissipation properties.

Lateral extent and thickness of the repository formation may also be important because of enhanced containment capability provided by high sorptive properties of certain rock types. The greater the pathway, the greater the opportunity for radionuclide retardation by sorption processes discussed in Section C-3.1.2.

The minimum thickness or lateral extent criterion may also be dictated by:

- The distance required to protect the repository from groundwater flow or to account for unknowns about the material's purity that could affect groundwater flow.

- The distance from the mined cavity to the extremities of the formation to minimize the potential that localized cracking or straining could open pathways beyond the desirable containment zone.

C-3.2.2.2 Depth of Repository Horizon

The necessary depth of a waste repository horizon will vary depending upon many factors. Those factors associated with the host rock are:

- Location of suitable host rock,
- Erosion and denudation potential,
- Local groundwater conditions,
- Depth of weathering,
- Depth of open fracturing,
- Ambient temperatures at depth, and
- Short- and long-term mine stability considerations.

The actual depth criteria for a specific location will be site-related. Mined openings in shale could possibly be extended deeper than in salt because of the differences in physical properties. Brittle rocks, such as limestone, granite, and basalt, can undoubtedly be mined more easily at greater depths than would be practicable for shale or salt. These "engineering" limitations must be considered in the context of all site selection criteria. The most common depths for disposal may range from about 300 to 1500 meters, but shallower or deeper rocks may be suitable under certain site conditions.

C-3.2.2.3 Uniformity and Homogeneity of Geologic Media

All other factors being equal, rocks having a high degree of uniformity and homogeneity are superior for emplacement of radioactive wastes. The major reason for this conclusion is that greater accuracy of all conclusions from investigations and analyses may be achieved when site variability is reduced; i.e., the chance of missing an important factor is greatly reduced when site conditions are uniform. Nonetheless, sites that are neither uniform nor homogeneous may be suitable, if other conditions are particularly attractive.

The acceptable degree of inhomogeneity will vary, depending on such factors as the nature and characteristics of rocks or minerals included in the host rock and the location of potential pathways from the repository area. For example, it is common to find thin partings of shale, several meters apart within a thick limestone or salt stratum. The nature and location of these partings are important in mining. When widely spaced, they can be used during the mining to form a smooth and continuous roof. On the other hand, depending on the nature and spacing of the partings, they may contribute to roof instability, act as pathways for groundwater flow, or cause localized fracturing from differential expansion with temperature changes. Also, changes unrelated to layering, such as the occurrence of hydrated minerals, which can react adversely with waste products or be a source of water under high temperature conditions, may be significant. The importance of inhomogeneity must be decided on a site-by-site basis.

C-3.2.2.4 Nature of Overlying, Underlying and Flanking Strata

It is implied throughout the above paragraphs that acceptable conditions of the host rock will often depend directly on the surrounding environment. Accordingly, it is necessary to determine the detailed stratigraphy of the formations overlying, underlying, and flanking the repository. For example, thick beds of shale and/or other plastic-behaving rocks may be desirable adjacent to the repository formation if they are: relatively impermeable to circulation of groundwater; have high radionuclide retardation potential; and can be deformed without fracturing. Probably the least desirable rocks surrounding the repository are highly fractured and contain large quantities of groundwater.

C-3.2.3 Structural Geology

The subject of structural geology characterizes and describes features of rock materials that have been created by earth movements. In addition to descriptions of existing conditions, structural geologic studies provide valuable interpretative information that can greatly facilitate prediction of future events. The three major structural geologic aspects of interest for this study are:

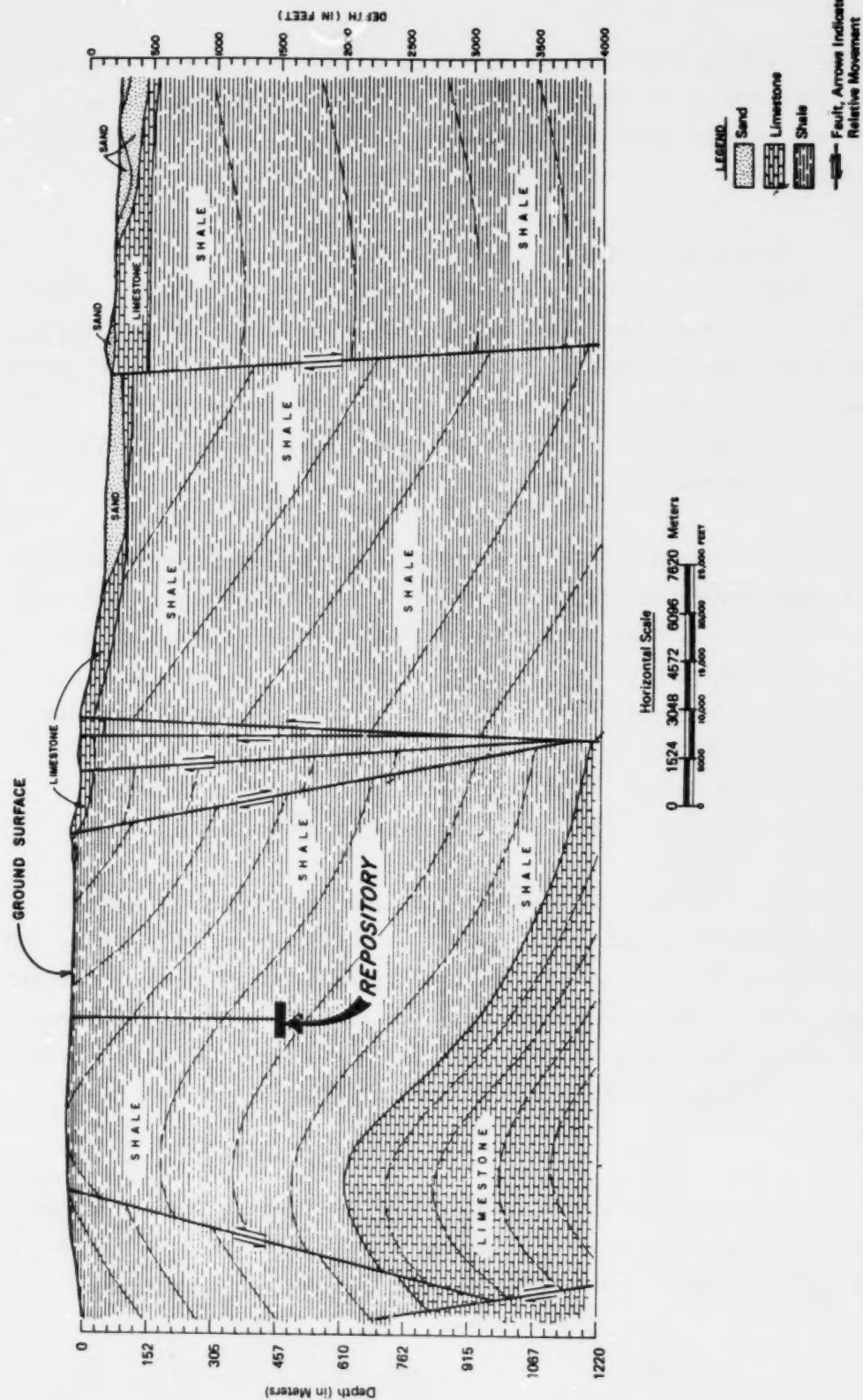
- Folding of strata, which describes the spatial orientation of bedding;
- Fractures including primarily joints and faults. occurring in otherwise continuous geologic materials.
- Major plastic flow of materials, such as diapirism in salt domes.

C-3.2.3.1 Folding

The degree to which strata have been folded is important because of the resulting inclination of bedding and fracturing of the rock where it has been forced into a curved configuration. Figure C-2 presents an example of moderately folded shale and limestone. In this cross section, the rocks have been deformed into an anticlinal structure (a fold which is convex upward).

From repository design and operations viewpoints, the preferred dip of rocks (or strata inclination) should not exceed a few degrees. A nearly horizontal alignment facilitates hauling the excavated rock and subsurface transport of the HLW materials. This consideration can be of lesser importance for very thick, relatively homogeneous rock where the opening does not necessarily have to follow the stratigraphic horizons or beds.

Sudden and frequent reversals of dip are not only undesirable for design and operations, but the abrupt folding that caused the reversals may have created undesirable secondary permeability (Section C-3.1.1) in the rock. This is particularly true for hard and brittle rocks in which stress conditions at the time when folding occurred have usually increased the frequency of fractures in the formation, resulting in a significant increase in the potential for groundwater movement. For plastic-behaving rocks like salt, deformations accompanying moderately



Source: D'Appolonia Consulting Engineers Inc., for Arthur D. Little, Inc.

FIGURE C-2 RADIOACTIVE WASTE REPOSITORY CONCEPTUAL MODEL IN TECTONIZED SHALE

to steeply dipping beds may have little or no effect on water movement as fractures are healed with time. The resistance to fracturing of shales lies between salt and the more brittle rocks, such as limestone and basalt.

C-3.2.3.2 Joints and Faults

Joints and/or faults are common geologic features in most rocks. Joints are fractures in rock that have experienced no relative movement between the two sides, whereas faults are fractures along which relative movement (vertical or horizontal) has occurred. The discussion in this section deals with existing fault conditions, while Section C-3.2.5 discusses the potential for future faulting.

Joints are very common in all types of lithologies, ranging from volcanic rocks to unconsolidated sediments. Joints are most commonly oriented approximately perpendicular to bedding, but they can be oriented in any direction, depending upon the stresses or deformations that caused the rock to crack. They may be open, having a space large enough for water to pass, or they may be closed or filled with impermeable secondary minerals, such as calcite or quartz. The open type are of primary concern because of their ability to transmit groundwater.

Some degree of jointing will usually be present in relatively shallow shales, limestones, basalts, and granites. The frequency of jointing and the probability of joints being open usually decrease with depth. Their characteristics at any depth must always be determined by boreholes and field or laboratory testing. Sometimes open joints are found at depths equal to or greater than those typically considered for HLW repositories.⁽⁵⁾

Ideally, a potential repository area should be virtually free of faults, whether active movement has occurred recently (within the past 20,000-30,000 years) or not. Minor, recently inactive faults in an area

might be acceptable if it is possible to show that (1) the faults were due to some tectonic force that is not likely to reoccur, (2) all disturbance was outside of the actual disposal host rock or, (3) the rocks are sealed against potential groundwater flow.

Any existing faults that have occurred in the recent geologic past may be sufficient cause to make the site unacceptable. At a minimum, recently active faults require extensive investigations to ensure that the host formation will not lose significant containment capability through faulting during the facility's designed life.

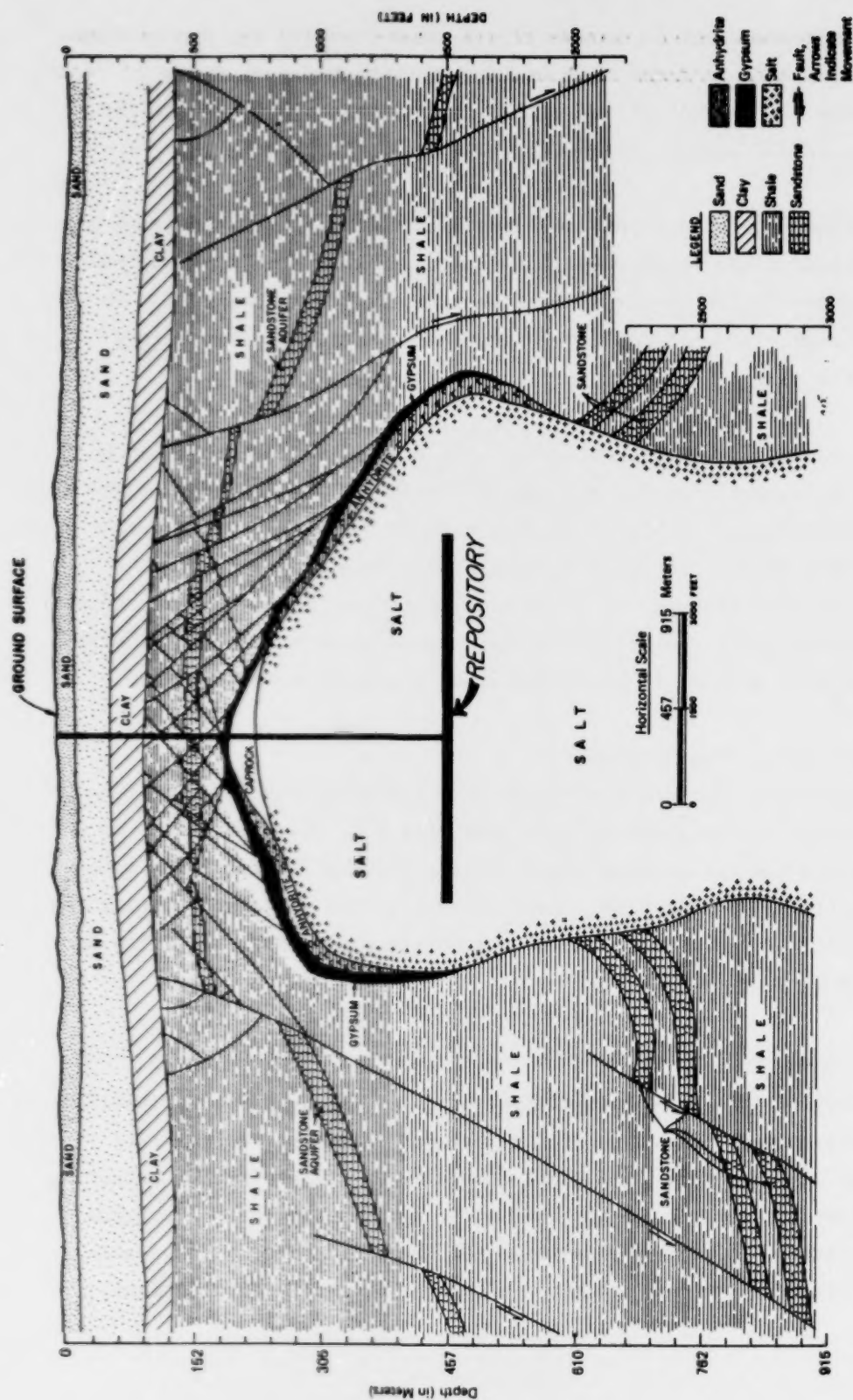
Major lineaments (lengthy features on the earth's surface that may be straight or slightly curved and can be observed on aerial photographs and satellite imagery) also fall within the broad category of fractured rock. These lineaments may represent areas of increased jointing in subsurface rocks or they may represent major faults. Unless such features can be positively shown to have no effect on containment, they should be avoided in selecting areas for waste repositories.

C-3.2.3.3 Major Plastic Flows

Major structural features associated with upward plastic flow of geologic materials considered in this study are salt domes formed by an uplift mechanism called diapirism. Figure C-3 illustrates a geologic cross section of a salt dome. Note how the intruded strata flanking the salt are folded and faulted due to the uplift pressures. Diapirism is discussed in Section C-3.2.5.

C-3.2.4 Hydrogeology

The concept of investigating and designing any facility with scientific confidence in its integrity for time periods of perhaps hundreds of thousands of years adds an unusual dimension to all aspects of selecting a site based on geologic characteristics. The impact of this extreme time constraint is possibly greatest for the consideration of groundwater movement. For typical water supply or dewatering evaluations,



Source: D'Appolonia Consulting Engineers, Inc., for Arthur D. Little, Inc.

**FIGURE C-3 RADIOACTIVE WASTE REPOSITORY
CONCEPTUAL MODEL IN SALT DOME**

water flow is considered relative to its volume per minute, day, or year. However, over time periods of thousands or hundreds of thousands of years, water flow that is imperceptible on a per-day or even a per-year basis may be significant. In other words, rock permeability is a relative concept and the aspect of time available for flow to occur is significant.

For a repository, a small volume of flow over each one thousand or ten-thousand-year period could be important, if it could transport unacceptable levels of radioactive materials from their repository setting. To evaluate the required detailed hydrogeology will require extensive investigations. Relationships must be established among such factors as the following:

- Regional groundwater conditions,
- Local (at the site) groundwater conditions,
- Intrinsic hydrogeologic properties of the disposal zone and associated strata.

Elements of intrinsic hydrogeologic properties were described in Section C-3.1.1, and associated geochemical characteristics for retarding radionuclide travel were addressed in Section C-3.1.2. These basic data are not repeated in the following broader description of hydrogeology.

C-3.2.4.1 Regional Hydrogeology

For this study, regional hydrogeology relates to the existence and extent of all aquifers in the vicinity of a possible repository that contain water that could flow above, below, or beside the repository given any natural or man-caused event during its designed containment life. Depending on site location and the degree of detail of the investigation, the study of regional hydrogeology can cover an area from several hundred acres to hundreds of square miles. The area of interest probably will be greatest where extensive sedimentary deposits are considered as the host rock.

Important regional questions follow:

- Where are any aquifers and aquicludes located in relation to the host formation?
- Where is the recharge area and what is the recharge mechanism for each aquifer?
- Is there natural flow in each aquifer system? If so, what is the gradient causing flow and where are the discharge locations?
- Where and why is there flow between separate aquifers, including consideration of boreholes for water supply, gas and/or oil extraction, or mineral investigations?
- What is the quality of groundwater in each aquifer; is it presently being used or does it have potential for exploitation by man for irrigation, drinking, or other purposes?
- Are there mechanisms that could provide a connection between any aquifers and the disposal formation, including geologic events that may occur at significant distance from the repository?
- How will extreme changes in the total hydrologic (precipitation-runoff-infiltration-evaporation) cycle in the region during the very long containment period affect the regional hydrologic system? Both extremely wet and extremely dry periods must be considered.

Investigation of these regional conditions must always be carefully planned and executed. The program must then be continuously modified as new data are evaluated and new methods are proved. It is not practical in a short period to determine regional hydrogeology by specific field investigations. Conclusions must be developed based on:

- The large volume of stratigraphic and structural geologic literature available,
- Logs of wells drilled in the area,
- Geologic data obtained from aerial photographs and imagery interpretations along with field verification of important observations and evaluations, and

- Site investigations at selected areas throughout the region of interest.

Sometimes the data will be sufficient to rule out completely large areas otherwise considered as potential repository sites. More often, however, the regional study will identify important factors for more detailed local and site-specific studies.

C-3.2.4.2 Local Hydrogeology

The study of local hydrogeologic conditions brings together the regional character of aquifers and aquicludes and the intrinsic properties of the host rock. Such a study is used to determine if:

- Groundwater can reach the extremities of the host formation,
- Groundwater can pass through and escape from the host rock with the potential for transporting radionuclides, and
- Any radionuclides can leave the repository and go to an aquifer that may eventually transport them out of the containment zone.

The key evaluations will usually be to determine if: (1) there is any type of gradient (usually pressure gradient) that could move groundwater through the containment area; (2) material permeabilities are such that flow will be significant; and (3) benefits or detriments may be realized from the geochemistry of the system. In the final analysis, the most important hydrogeologic consideration is whether or not groundwater can transport an unacceptable level of radionuclides to the biosphere.

Local hydrogeologic studies begin with the regional information and use specific information derived from boreholes, such as pressure measurements, water level measurements, and water quality and temperature analyses.

Hydrologic and geochemical modeling analyses and other site-specific investigations appropriate to defining existing and potential flow conditions are also employed. Detailed knowledge of stratigraphy and structural features at the repository site (Sections C-3.2.2 and C-3.2.3) are necessary to the success of any local hydrogeologic study.

Special note is made that the plugging of open boreholes or sealing around installed piezometers and other instrumentation will be very important for any repository investigation. New methods for sealing between aquifers encountered in a single borehole are being developed for this purpose.

C-3.2.5 Tectonics

This general site selection category deals with significant earth movements (geologic events) that could disrupt the repository in some manner that would reduce containment. The discussion is separated into four subtopics:

- Tectonism, which generally includes all types of movements of the earth's crust. For repositories in rock formations, the definition can be limited to (1) faults, which cause abrupt disruptions at the fault displacement and increased brokenness at considerable distances and (2) less abrupt movement caused by folding, which can increase the brokenness of rock, change erosion patterns and rates, and generally modify regional or local hydrologic systems.
- Seismicity, which for repository site selection purposes is related to (1) the frequency of earthquake activity that occurs in an area and (2) the magnitude of movement that occurs as the result of an earthquake. The first relationship is closely related to the faulting of tectonism and is not discussed separately. The second relationship will have significant importance on the engineering design of above-ground and underground facilities during the relatively short operating life of a repository, but little influence on containment for the very long period after backfilling.

- Diapirism, which for this study relates mainly to the creation of salt domes by the upward flow of this plastic rock from a deep-bedded deposit.
- Volcanism, which refers to processes whereby magma (molten rock) and its associated gases rise into the earth's crust and are extruded into the atmosphere and onto the earth's surface where rocks are formed as the magma cools.

C-3.2.5.1 Tectonism and Seismicity

Tectonism in the context of this study is related to fault disruptions that actually cause abrupt movement between two portions of the earth's crust and/or significant folding of the crust with a resulting increase in the brokenness of rock formations. Basic questions requiring consideration in site selection are:

- How often, if ever, are significant movements to be expected?
- What will the magnitude of movement be?
- Where will the movement be located?
- What are the potential impacts on the containment capability of a repository?

The second and third questions overlap significantly with seismology or seismicity investigations that make possible predictions based on evaluation of past conditions and the geologic setting. For purposes of this study, discussions of tectonism and seismicity are combined.

An earthquake is a natural phenomenon in which significant rapid and/or violent movement of the earth's surface occurs. Because earthquakes have occurred in most areas of the United States during the relatively short period of recorded history (approximately three centuries), they are of immediate concern in selecting a repository site, particularly in light of the desired long containment time period. The magnitude of earthquake disturbance and the location and depth of movement are quite variable for different parts of the country. Even in a given area, the potential for

rock movement or fracturing as a result of a nearby earthquake will depend on such local factors as stratigraphy (Section C-3.2.2) and stiffness or elasticity (Section C-3.1.4) of the affected rocks.

The principal tasks in selecting a suitable site will include:

- Quantification to the extent possible of the earthquake and faulting history of the site region.
- Establishment of a geologic rationale for the historical behavior and using this as a basis for predicting probable future activity.

Areas of high seismic or faulting risk, such as portions of the west coast of the United States, will have very low acceptability and areas of low risk will appear favorable if other desirable geologic characteristics are present. It is possible that almost any area may be acceptable from the seismicity viewpoint where it can be reasonably shown that movements at or near the repository cannot occur with sufficient magnitude to reduce containment capability.

The prevailing geologic explanation for regions of the world within unusually high earthquake activity is centered on two concepts--continental drift^{(15) (16)} and sea floor spreading⁽¹⁷⁾ which have been unified in the theory of plate tectonics^{(18) (19)}. Continental drift was proposed in 1912 by German meteorologist Alfred Wegener⁽²⁰⁾ who argued that the earth's crustal rocks could flow laterally if they could flow vertically. Thus, he postulated that all the continents had been joined 200 million years ago, after which they had broken apart. Alternative views to Wegener's were postulated by Wilson⁽¹⁶⁾ and earlier by DuToit⁽²¹⁾ as well as others to dispose of various problems that Wegener's model raised. Within the past 20 years the theory of continental drift was placed on a firmer foundation by development of the sea floor spreading hypothesis originally proposed by Dr. Harry Hess of Princeton.⁽²²⁾ This concept of sea floor spreading was substantially confirmed by the work of F.J. Vine and D.H. Matthews of the University of Cambridge.⁽²³⁾ Magnetic data from the vicinity of the oceanic ridges confirmed that sea floor spreading has been going on during the past 200 million years at rates of between 2 and 18 centimeters per year. That is, crust has been more or less continuously created during the past 200 million years, accounting for

most of the present areas of the oceans. Since evidence suggests that the earth has not expanded more than 2% during the last 200 million years, there arises the implication that crust has also been destroyed. Plate tectonic theory accounts for this implication.

The significance of plate tectonics to the waste disposal issue lies largely with the seismicity/earthquake phenomena related to plate movement/boundaries. Dewey⁽¹⁸⁾ points out that "Most earthquakes occur in narrow zones that join to form a continuous network bounding regions [the plates] that are seismically less active." Figure C-4 illustrates the distribution of these zones and indicates the larger plates by name.

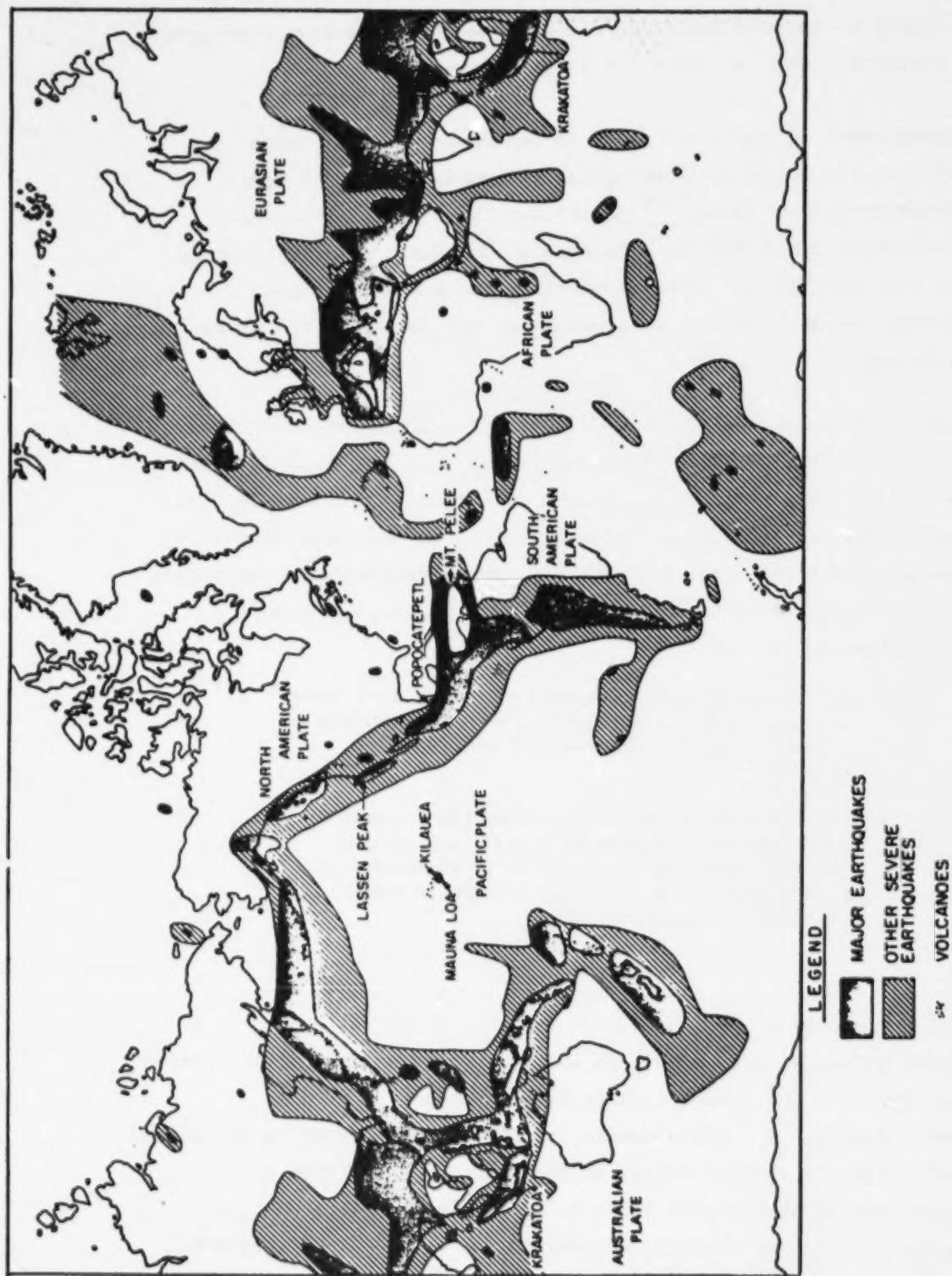
The concept of plate tectonics also relates to tectonism, which deals with the structural folding of geologic formations over geologic time. Active folding or curving of a geologic formation is not usually a repository design consideration, because even if in progress, it occurs very slowly. Folding, however, may be a consideration for a repository requiring controlled containment for geologic periods of time.

Major considerations are:

- Can slow geologic deformations occur that have the potential to cause the rocks to fracture (crack), opening avenues for radionuclide escape?
- Are the host rocks and surrounding formations sufficiently flexible (e.g., salt and shale) and self-healing (e.g., salt) to withstand slow movements without any change in their containment capabilities?

C-3.2.5.2 Diapirism

Diapirism refers to the process in which geologic structures are formed as the result of the upward plastic flow of rock from deep to shallower levels. For the selection of HLW disposal sites in the United States, diapirism relates mainly to the salt domes in Texas and Louisiana and in the Paradox Basin of Utah. These salt domes formed as a result of tectonic downwarping of a bedded salt deposit, after which upward plastic flow occurred through overlying strata. The conceptual geologic model shown in Figure C-3 and discussed in Section C-3.4.2 illustrates this type of geologic setting.



Based on: H. Williams, Scientific American, November 1955 (24) and J.F. Dewey, Scientific American, May 1972.

FIGURE C-4 REGIONS OF THE WORLD IN WHICH MAJOR EARTHQUAKES AND VOLCANOES OCCUR

Geologic considerations for a salt dome are significantly different than for bedded salt or other rock types. The domes often have a lateral extent of only about one or two kilometers, but are continuous vertically downward for several kilometers. The selection of a dome site will largely be dependent upon the desirable lithologic properties of the salt (Section C-3.2.1). Stratigraphy (Section C-3.2.2) and regional and local hydrogeology (Section C-3.2.4) will be less important for evaluating basic containment potential.

A site selection criterion for any salt dome will be the current status of any local or nearby diapiric activity, or the potential for renewed activity during the design life of the repository.

Another criterion of lesser importance will be the assessment of possible diapiric formation in regions that have no historic geologic evidence of such structures. This consideration is related to the possibility of disruption of the repository by upward flow of material. The prerequisites for such an event are a plastic-behaving rock buried at great depth (up to 8000 meters or more),⁽²⁵⁾ under the required temperature conditions (at least 300°C)⁽²⁶⁾ and beneath the proper geologic strata. An early check on the potential for the necessary combination of factors will likely rule out the possibility of diapirism occurring at most sites which do not now have any history of such events.

C-3.2.5.3 Volcanism

Areas in which volcanism might reasonably be expected to occur within the next million years should be avoided in selecting a HLW repository site. Obviously, the high degree of violent activity associated with a volcanic eruption would affect the containment capability of a repository stratum in close proximity to the volcano.

Two basic types of landforms can result from volcanic activity.⁽²⁷⁾ In some areas, molten rock has been extruded through extensive fissures in the earth's surface and has built large plateaus (fissure eruptions). In other regions, molten rock escaped from within the crust through vents; around these vents ejected materials built up into landforms called volcanoes.

A volcano is considered active if it is presently erupting or is expected to do so. If it is not active but may well resume in the future, the volcano may be termed dormant. If no signs of activity are present and no activity is anticipated in the future, the volcano is considered extinct.⁽²⁸⁾

Ninety-eight percent of all active volcanoes are located along plate boundaries where significant earthquake activity is also concentrated. The Western quarter of the United States is such a zone and includes Lassen Peak in California, which was last active in 1917.

C-3.2.6 Erosion and Denudation

HLW repositories must be located in areas and at depths where waste will not be exposed as a result of erosional processes. Major erosional agents include rivers and streams, landslides and other mass movements; rain, which erodes by impact and surface flow; frost action, which loosens surface materials; chemical weathering of surface materials; wind, which erodes by deflation and sand blasting; and glaciers, which erode by grinding and plucking. For this study, the important factors are erosion by river flow and denudation by glacial action.

Erosion is a continuously occurring process, the immediate results of which are nearly imperceptible, except in the case of mass movements. However, over geologic time periods erosional effects can be quite perceptible--these highly evident results that can have marked effects on landforms are connoted by the term denudation. In evaluating erosion

and denudation in the overall assessment of the suitability of a site for a HLW repository, careful consideration must be given to present erosion rates and the potential for substantial increases in erosion rates over long time periods.

Rates of erosion vary according to specific topographic location, rock hardness, rainfall, evaporation rates, type of vegetation and other factors. As mentioned in Section C-3.2.5, factors such as tectonism can cause broad uplift of land surfaces, which can lead to radical changes in erosion rates. For example, the Grand Canyon portion of the Colorado River has eroded to a depth of 2000 m during an estimated one million year period.⁽²⁶⁾ The overall rate of erosion in major river basins of the United States averages about 60 m/million years. Erosion rates range from about 40 m/million years in the Columbia River basin to about 165 m/million years in the Colorado River basin.⁽¹⁴⁾ Since repositories will not be placed in areas where major ground movements or river erosion can be hypothesized, erosion will usually not be a major site selection factor.

For many sites, potential glacial activity may be the most important factor for evaluating erosion or denudation potential. Glaciers often deepen and widen valleys previously formed by streams; where tremendous thicknesses of ice occur, valleys may be deepened to great depths. Examples are the fiords of Norway, which are estimated to have been incised to depths of 1300 m by glacial activity.⁽²⁷⁾

Four glacial periods have occurred in the northern United States during the past million years. The occurrence of one or more additional glacial periods during the next million years must be considered as a potential factor in the development of an HLW repository.

C-3.2.7 Mineral Resources

The attractiveness of many sites will be affected by past or potential future interest in a mineral resource. Mineral resource considerations

include surface and underground mining of solids, solution mining, and the withdrawal of subsurface fluids, such as oil, gas, and water. Each identified resource requires consideration of any past exploration or recovery activities and an assessment of the potential for exploitation during the life of the repository.

The exact nature and locations of all existing mines and wells near any site must be identified and their effect on the confinement ability of the site must be analyzed. This analysis should include effects of subsidence from mining or pumping, the extent and location of induced permeability, caused by hydrofracturing (fracturing rocks by injecting water under high pressure), and the extent of exploration testing (e.g., borings, test wells, underground blasting for geophysics) that could have an impact on containment potential. At some locations, the measurement of existing conditions may present the greatest difficulty in certifying a site's suitability.

Any potentially valuable mineral in the formations above, adjacent to, or below the host rock presents two selection considerations:

- The existence of the mineral raises the possibility that man may disturb the containment in the distant future by exploring for or extracting the mineral, and
- A necessary and valuable resource may be lost to man because of its location close to a repository.

C-3.3 CATEGORIZATION AND RANKING OF SITE SELECTION FACTORS

The discussion sequence in Sections C-3.1 and C-3.2 ranged from the detailed examination of the intrinsic properties of the host rock to the larger-scale evaluations of geologic settings and finally, geologic events. A logical procedure for locating and investigating potential repository sites moves in the opposite sequence, from considerations of the large scale settings or events to intrinsic properties. The following sections present an appropriate starting procedure for most HLW repository site selection programs.

C-3.3.1 Categorization of Factors

Table C-6 summarizes geologic information categories that are pertinent to selecting a site for a HLW repository. The categories are arranged to suggest a logical investigative sequence (top to bottom in the table). The first stage of the investigation involves assessing the potential for the occurrence of significant geologic events (including those associated with mineral resource exploitation or loss) in candidate regions. Areas with a high probability for such an event should be eliminated from consideration; no further work would be performed relative to those sites.

The second phase of the investigative sequence would concentrate on the evaluation of the geologic setting for the remaining sites. This phase of the study will (1) eliminate other sites on the basis of negative regional or local geologic conditions, and (2) better identify the most promising specific locations within acceptable areas. It is important to begin a relative ranking of sites at this stage in order that further detailed studies can be tailored to the most promising areas in an appropriate manner. Each element within the geologic setting category (Table C-6) should be fully addressed during the ranking process.

The final step in site selection will be to gather specific local data for the best site(s). These investigations must provide detailed stratigraphic, structural, and hydrogeologic descriptions, as well as quantitative information about the host rock's lithology and intrinsic properties. These final data, with appropriate analyses and evaluations, permit the identification of site limitations, special design, operation and/or abandonment requirements, and allow final site ranking based on geologic considerations. Final site selection will require that the geologic results be evaluated in conjunction with other selection factors, such as transportation routes, and socio-economic considerations.

TABLE C-6
GEOLOGIC INFORMATION CATEGORIES FOR
SELECTION OF A SITE FOR HIGH
LEVEL RADIOACTIVE WASTE DISPOSAL

INFORMATION CATEGORIES	ELEMENTS IN EACH CATEGORY
Geologic Events	Tectonism Seismicity Diapirism Volcanism Erosion and Denudation Glaciation Mineral Resources
Geologic Setting	Lithology Thickness and Lateral Extent of Host Rock Depth of Host Rock Uniformity and Homogeneity of Host Rock Nature of Strata Surrounding Host Rock Regional and Local Folding Joints and Faults Major Plastic Flows Regional Hydrogeology Local Hydrogeology
Intrinsic Properties	Primary Permeability Secondary Permeability Porosity Ion Exchange Capacity Mineralogy and Solubility Pore Water Chemistry Water Content Thermal Properties Engineering Properties

C-3.3.2 Ranking of Factors

It is not practical to provide a quantitative ranking of geologic site selection factors presented in this study, because the emphasis required for each factor will vary, depending upon the host-rock type and the regional and local geologic setting. However, it is reasonable to discuss a logic for ranking the factors to provide a starting point for any specific study.

The importance of any of the geologic event factors will depend on the area being investigated. Not all geologic events will have the same likelihood of occurring in specific geographic/geologic regions. Example questions regarding geologic event possibilities follow:

- Does the region include any sites near geologic plate boundaries (Section C-3.2.5) where the occurrence of volcanic action or major faulting is highly likely?
- Does the area contain conditions of known, active diapirism?
- Is it probable that future glaciation stages will cover the area?

Part of the overall ranking of events should consider the probability that certain events will occur in some areas. For example:

- Mineral resources will be a significant consideration for any area.
- Major faulting or folding will always be a consideration, but catastrophic earthquakes will not be a significant consideration everywhere.
- Glaciation, diapirism, and volcanism will be more geographic and can be quickly ruled out for certain areas.

Again, it is emphasized that the very long containment requirement dictates that each investigation must initially consider all possible geologic events.

The following listing suggests a sequential procedure for considering geologic setting factors:

- Does the proper lithologic formation exist to function as a host rock for a HLW repository?
- Are there any structural features that may eliminate the site's potential?
- Do the stratigraphic and structural features present any unusual groundwater characteristics that must be considered?
- Are the structural and hydrogeologic conditions suitable for containment?

Ranking of the relative importance of intrinsic properties during the detailed site investigation phase will also be highly dependent on individual site conditions. A reasonable sequence of considerations for evaluating intrinsic properties follows:

- Does any mechanism exist such as groundwater flow that could transmit radionuclides?
- Are there chemical or thermochemical properties of the waste that could accelerate exposure of radionuclides to the host formation and cause an undesirable reaction with the host formation?
- Are there geochemical properties of the host formation and surrounding geologic media that could retard the migration of radionuclides if a transport mechanism does exist?
- Can the host formation and geologic surroundings dissipate heat from the waste and are there any potentially serious limitations of the host formation at high temperature?
- Can the host material be engineered to obtain the required operating and abandonment characteristics?

C-3.4 CONCEPTUAL GEOLOGIC MODELS

The conceptual models discussed in this section are illustrated in

Figures C-2 through C-7. The models are general and broadly represent various repository settings; nature is vastly more complex. Thus, the models are provided only as a basis for discussing site-selection factors for deep geologic disposal of HLW.

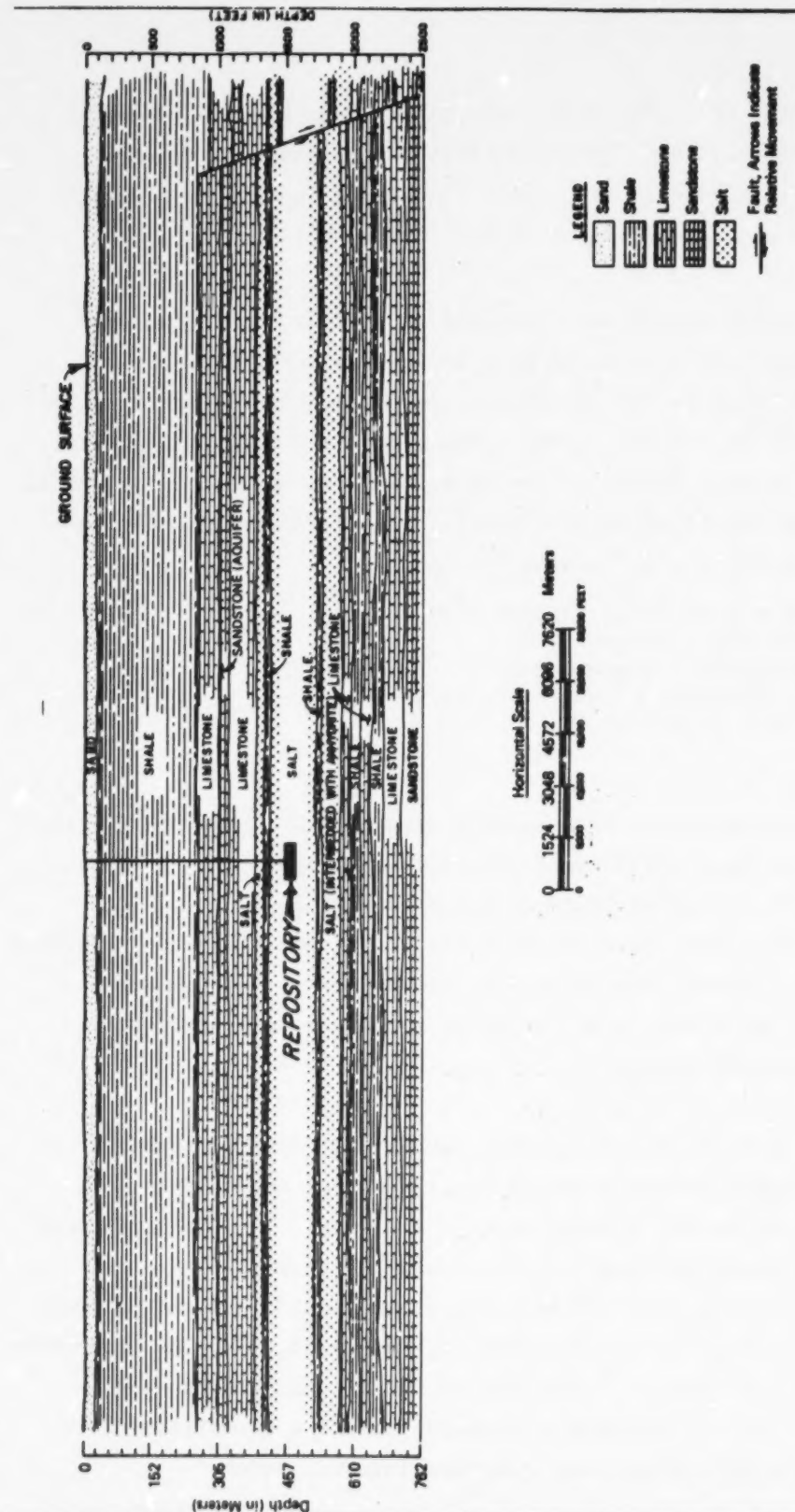
All of the conceptual models are presented at similar scales and each hypothetical repository is located at a depth of about 500 meters. This depth is used solely for descriptive purposes, but is generally considered reasonable for many sites. The depth at any given site will, as previously discussed, depend on the lithologic, mechanical, and chemical properties of the host rock and the general geologic setting. The conceptual models chosen for discussion follow:

- Tectonized Shale (Figure C-2)
- Salt Domes (Figure C-3)
- Bedded Salt (Figure C-5)
- Non-Tectonized Shale (Figure C-6)
- Basalt (Figure C-7)

C-3.4.1 Bedded Salt

Salt deposits are considered as a leading candidate for HLW repositories. The conceptual geologic model for a repository (Figure C-5) is an essentially horizontal, relatively uniformly bedded salt deposit. At many locations, the salt body would be characterized by an interbedded sequence of salt (halite), gypsum, anhydrite, limestone, and in most cases, shale. Only two shale layers are shown in the conceptual model to illustrate some interbedding.

As discussed in previous sections, salt has several very attractive physical and chemical characteristics for a disposal site. Salt possesses a very high thermal conductivity on the order of 15 millicalories per centimeters-second degrees C. This property, coupled with a large lateral extent of a bedded salt formation, would be a viable mechanism for dissipating the heat generated by the waste. The low permeability (often much less than 10^{-6} centimeters per second) and the low interstitial porosity (less than 1 percent) provide a significant barrier to fluid movement other than localized internal water.



Source: D'Appolonia Consulting Engineers, Inc., for Arthur D. Little, Inc.

FIGURE C-5 RADIOACTIVE WASTE REPOSITORY CONCEPTUAL MODEL IN BEDDED SALT

The presence of shale interbeds could retard radionuclide transport. Although shale has a moderate interstitial porosity on the order of 5%, it has a low permeability (usually less than 10^{-4} and often as low as 10^{-6} to 10^{-11} cm/s) and a high sorption capacity, which would inhibit the migration of radionuclides from the repository horizon. If the shale contained hydrous mineral phases, a potential negative impact results from the release of water by heat generated from the waste decay.

The conceptual model shows that the salt-shale sequence is overlaid by a limestone unit. If relatively unbroken and without unfilled solution channels or joint systems, the limestone should have a low permeability to retard radionuclide migration to the overlying sandstone aquifer.

The sandstone aquifer shown overlying the repository is postulated to have a relatively large lateral extent. Often such a layer can transmit sufficient water for industrial and/or municipal uses. The aquifer is shown at a depth, however, that would probably be uneconomical for farm and domestic use, especially with the presence of a thick, water-bearing sand near the surface. Since the region in which this model is postulated is characterized by flat lying rocks, the sandstone aquifer will not crop out for at least tens of kilometers.

The sandstone aquifer in the conceptual bedded salt deposit is overlaid by a thick shale unit with very low permeability. This unit represents a significant barrier to radionuclide migration. As a result, waste contamination to the biosphere could occur only where the sandstone crops out, where the sandstone unit has been tapped for a water supply in an area where the shale is excessively broken, or where a borehole could allow vertical migration of fluids.

A very old, inactive fault is shown on the right side of the conceptual model. The fault cuts several of the stratigraphic units, including the salt and the sandstone aquifer. The effects of the fault on the aquifer could be that it would prevent horizontal radionuclide migration, but it might enhance vertical migration to at least the overlying shale unit. For the case shown, the clays and silts forming the shale were deposited after the tectonic activity that caused the fault had ceased.

A deep aquifer has been shown below the salt formation. This represents an additional potential escape route similar to the upper sandstone, but with much less potential for reaching the surface. The pressure gradient between the upper and lower sandstone aquifers also represents a basis for establishing whether water migration between the two, through the host rock, should be a design consideration.

C-3.4.2 Salt Domes

A conceptual model for a salt dome waste repository is shown in Figure C-3. The unique features of this model are:

- The intrusive nature of the salt into the surrounding geologic formations; and
- The large amount of faulting and fracturing present in the formations adjacent to and overlying the salt dome.

As discussed in Section C-3.2.5, salt domes in the United States are concentrated primarily in the coastal regions of Texas and Louisiana, and to a lesser extent in the Paradox Basin, located largely in Utah. The salt was originally deposited as a bedded sequence of halite, gypsum, anhydrite and shale. As a result of deep burial and differential pressures, the salt intruded into the overlying sediments, eventually forming salt domes.⁽²⁵⁾

The fracturing and faulting shown overlying the salt dome are the result of the forceful intrusion of the salt into the overlying sediments. These faults would not be expected to undergo movement once active dome emplacement has ceased. However, movements could occur from excessive withdrawal of subsurface fluids, such as water, oil, gas, or brine.

The dome in the conceptual model (see Figure C-3) is essentially circular in plan view, with the diameter on the order of 3000 meters. The salt dome is assumed to extend to a depth of at least 4000 meters, which is not uncommon for Gulf Coast and Gulf Interior Basin salt domes. The large vertical extent of the salt dome will assist in the dissipation of heat generated by radioactive decay.

The salt dome is overlaid by caprock consisting of anhydrite and gypsum, with minor amounts of calcite and sulfur. This type of caprock is common and generally has very low, if any, measurable permeability. Caprock originates from accumulation of the less soluble minerals of the salt body as its original top was dissolved during emplacement.

The salt dome penetrates, and is in part overlaid by, a thick shale formation. The shale will have the same type of mitigating effects on radionuclide migration as the shale discussed for the bedded salt model, except that its permeability may be greater as a result of fracturing, which occurred during the emplacement process.

The lower aquifers are both faulted and folded upward as a result of the doming process. Because sections of the aquifer have become effectively cut off from circulation and abut against the salt dome, stratigraphic traps can exist that may contain water, gas, and/or oil under pressure.

Conditions above a salt dome can vary greatly, depending on its depth and the last time that the dome was active in geologic time. These near-surface conditions should not be important at a properly selected dome because the principal geologic barrier against radionuclide migration in this model is the salt.

If it is hypothesized that some radionuclides escape from the salt, their pathway to the surface would be dependent on the site-specific conditions and where the breakthrough point occurs. As with the above discussion for bedded salt:

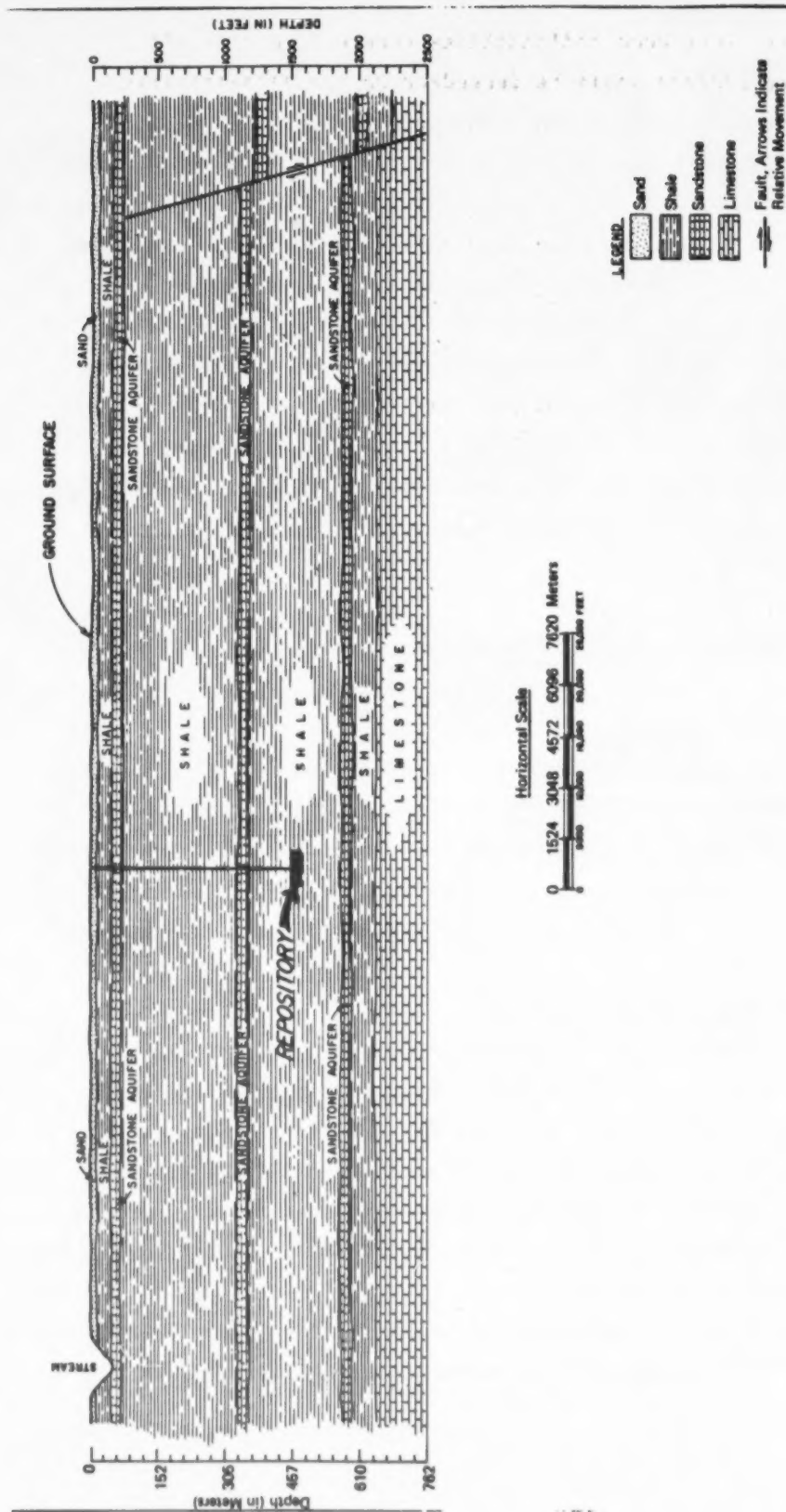
- Pathways to the surface could be greatly retarded by shale and clay layers that have low permeability and porosity and a high radionuclide retardation capacity.
- The number and location of sandstone zones, the amount and extent of brokenness of the surrounding rock and the potential for boreholes can contribute to increased pathways; again depending on site-specific conditions.

C-3.4.3 Shale

Some shales have properties that make them attractive for a waste repository. The geologic potential of a shale body for a waste storage site depends on dimensions of the formation, lithology, mineralogy, mechanical and hydrologic properties, structural and seismic history and the extent of drilling and/or mining.

The conceptual model for non-tectonized shale (Figure C-6) shows a relatively simple stratigraphic sequence composed of a limestone unit overlaid by a sequence of shale and sandstone strata. Each sandstone unit is considered to be an aquifer. The two lower aquifers have been offset by an old, inactive fault, which would limit the horizontal flow through these strata. The uppermost sandstone aquifer was deposited after movement of the fault had stopped, and it is considered continuous throughout the region. This upper aquifer is typical of conditions that often provide a primary water supply for both domestic and municipal use.

The principal factors for mitigating radionuclide migration from this first conceptual repository in shale are associated directly and almost exclusively with the thick shale units; i.e. low permeability, favorable sorption characteristics, and resistance to fracturing.



Source: D'Appolonia Consulting Engineers, Inc., for Arthur D. Little, Inc.

FIGURE C-6 RADIOACTIVE WASTE REPOSITORY CONCEPTUAL MODEL
IN NON-TECTONIZED SHALE

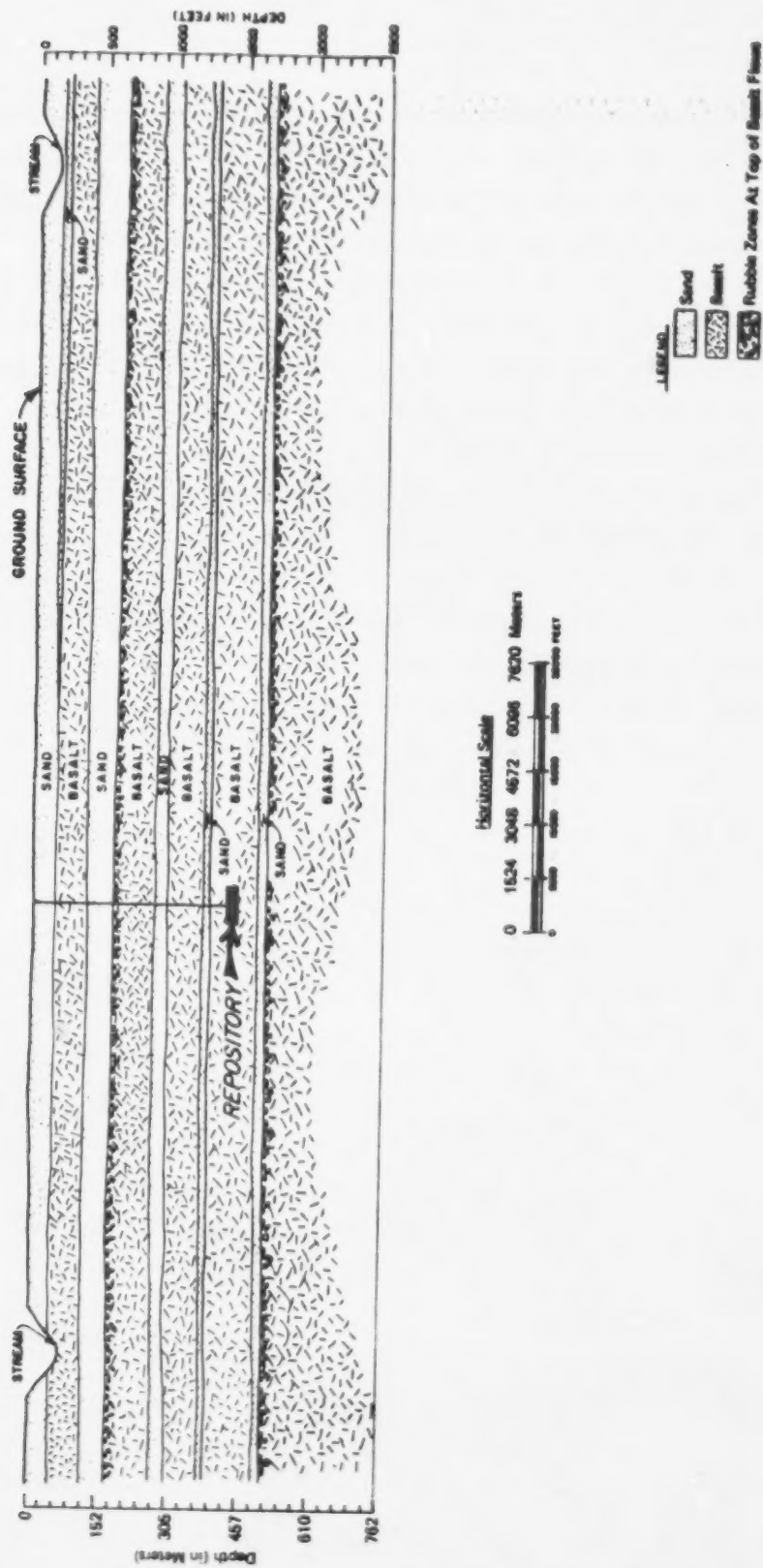
The second conceptual model in shale (Figure C-2) has been constructed in a tectonically-deformed shale unit, on the flank of a small anticline. The anticline is flanked by two high-angle faults. An obvious detrimental feature in this case is the presence of an extensive fault system that could present direct pathways for migration if groundwater were introduced into the repository. This site would be acceptable for a repository in a geographic location with an arid climate, if absence of water in the subsurface could be reliably predicted for the life of the repository. Even if water were to occur deep in the subsurface, the thickness and geochemical properties of the shale unit might be sufficient to retard nuclide migration. Substantial data would be required to demonstrate this conclusion.

Even without a widespread groundwater system, water could be generated in a shale repository through the dehydration of hydrous mineral phases of the rock. If this volume of water were small, no detrimental effects would be produced. Again, however, detailed studies would be required to demonstrate the actual volume of water that could be developed considering the temperature and size of the repository, and to evaluate any possible paths for radionuclides to be released from the repository along faults or fractured zones in the tectonically-affected rock.

C-3.4.4 Igneous Rocks (Basalt Model)

One of the most promising of the igneous rock types being considered for an HLW repository is basalt. The most attractive property of basalt appears to be its high strength for creating the mined cavities. Potential undesirable properties requiring careful investigation and evaluation are: basalt's relatively low thermal conductivity and its relatively high linear thermal expansion coefficient, which could either produce new fractures or expand existing fractures. Also, some basalts contain minerals that will decompose if heated in the presence of water. This is especially the case with pyroxenes and amphiboles.

A conceptual model of a repository in basalt is illustrated in Figure C-7. The thick deposits were extruded as sheet type (plateau) flows. These are the fissure eruption type volcanic deposits mentioned in Section C-3.2.5. The basalts shown are interbedded with fluvial sand deposits, which in some cases could also be lacustrine clay and silt deposits. In general, the clay and silt deposits would not be laterally extensive, but would form isolated lenses or pods between basalt flows. The sand units as well as rubble layers overlying some of the basalt flows often contain large amounts of water.⁽²⁹⁾ In one area, however, it is suspected that this water may not be connected to the surface; this is suggested by the age of the water, which has been dated as at least 40,000 years. If this condition can be proved, it may be possible to conclude that these "trapped aquifers" are likely not connected to a recharge area and would have little likelihood of transmitting nuclides to the biosphere. However, all considerations discussed for shales in Section C-3.4.3 would still be important on a regional and local basis for basalt.



Sources: D'Appolonia Consulting Engineers, Inc., for Arthur D. Little, Inc.

FIGURE C-7 RADIOACTIVE WASTE REPOSITORY CONCEPTUAL MODEL IN BASALT

C-4.0 ANALYSIS OF MIGRATION POTENTIAL

The primary objective of this subtask is to review the available information concerning, and quantify the potential for, migration of radionuclides through the geosphere. The greatest concern for such migration appears to be related to groundwater movement. The initial groundwater flow conditions (perhaps better described as deep subsurface water movement) at potential disposal sites are extremely important. The leaching characteristics of the high-level radioactive waste (HLW) are also important to the migration.

The approach used in the nuclide migration analysis involves several steps, including:

- (1) Review of the available data on waste leaching and of empirical and theoretical models describing leach rate,
- (2) Review of the available models of radionuclide transport in the geosphere, and
- (3) Selection and use of transport models for sensitivity analysis in order to identify the range of parameters important in nuclide migration.

The calculations for this report are shown to be applicable to the conceptual models for each of the five potential geologic repository types (summarized in Section C-3.4). A mathematical approximation of these geologic strata was then used for nuclide migration calculations. Details of this approach and the results are discussed in Sections C-4.3 and C-4.4.

C-4.1 MODES OF WATER CONTACT

The pathway used throughout this analysis involves nuclide migration to the earth's surface through geologic strata. In analyzing this pathway, two processes have been examined: flowing groundwater and diffusion through static groundwater. This section discusses the modes of possible water contact, the probable natural water flow rates, and the mechanisms whereby water might come into contact with HLW after the closing of a waste repository.

It will be important to have minimal or no natural water movement in the strata and interbeds of any potential repository. Since it is highly probable that in deep repositories there will be water-bearing strata

either above or below the potential repository area, water movement in the potential repository strata and interbedded strata can be expected to occur unless:

- (1) the repository formation has no permeability to water, or
- (2) the permeable strata above and below the repository are in hydrostatic equilibrium (i.e., water pressures differ only by the difference in elevation multiplied by water density).

There are probably no strata that satisfy the first criterion, but the combination of extremely low permeability and little, if any, hydrostatic imbalance could create a situation of virtually no flow in a repository. At least the water velocity could be below the usual threshold of hydrologic flow measurements.

Two kinds of hydrologic flow regimes were postulated to calculate the natural water velocity that might exist in a potential repository stratum. These calculations were made with a conventional two-dimensional hydrologic flow model. Although no specification of site type - bedded salt, salt dome, granite, basalt, shale, etc. - is intended, the example examined corresponds more obviously to a bedded formation than to other types. Nevertheless, the intrusive types of geology could still have water-bearing rocks above and below the potential repository area, thus, in both flow regimes, two aquifers were assumed to exist: one above and one below the potential repository strata. Two cases were investigated for each flow regime, illustrated in Figure C-8. The cross-section data applicable to each case are presented in Figure C-9.

In the first regime (Cases 1 and 2), flow in the two aquifers was assumed to be isolated until it reached the upstream point of entry to the region of interest in the model. At this point, a hydrostatic imbalance equivalent to 305 m of water (achieved by assuming an increase in upstream pressure in stratum 3 to 1403 m of water) was assumed to exist between the upper and lower aquifers (a condition that could occur at specific sites, although such sites would not be likely candidates for repositories). Depending upon the permeability of the repository of the

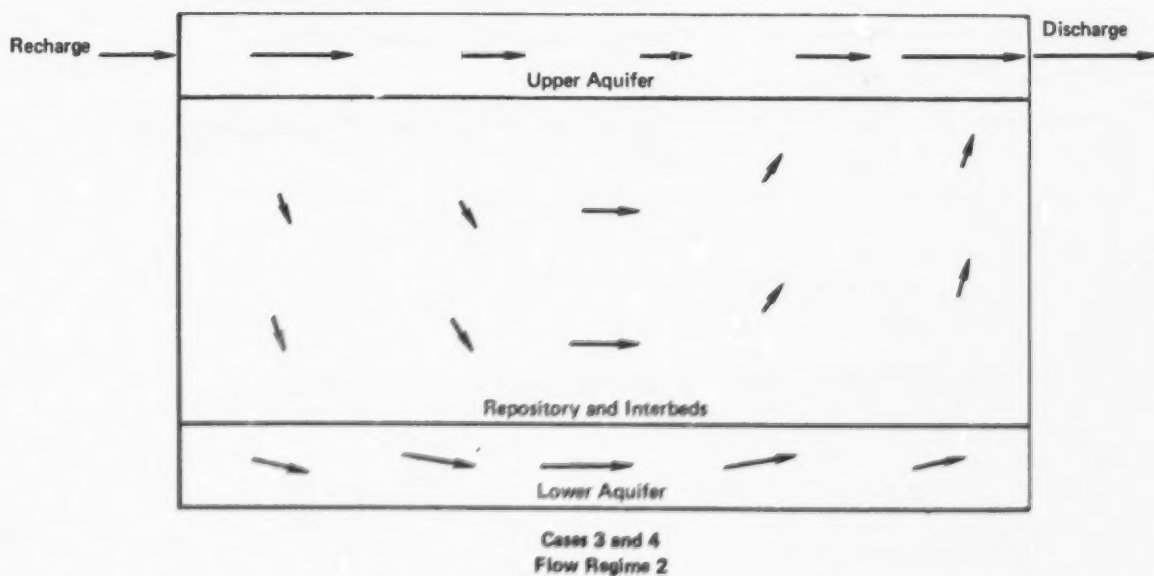
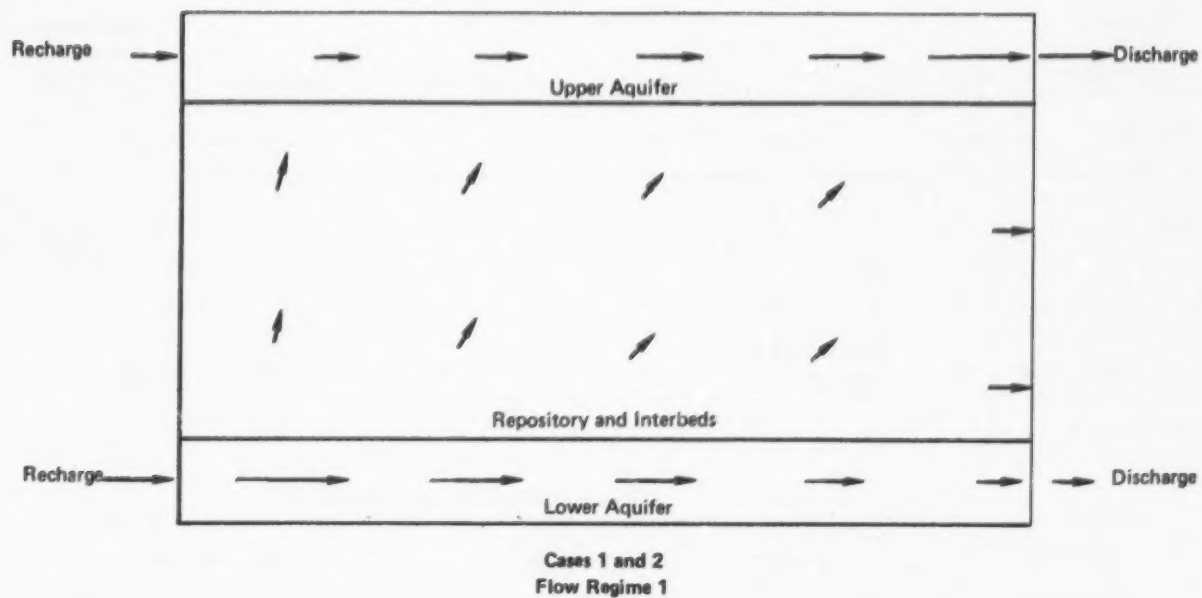


FIGURE C-8 TWO FLOW REGIMES

Upstream Potential (m of Water)	Base Case (No Vertical Flow)					Downstream Potential (m of Water)
	Stratum No.	Thickness, m	Permeability, cm/s	Porosity, %		
762	→ 1	Aquifer	30	10^{-5}	15	→ 0
930	→ 2	Repository and Interbeds	305	10^{-7}	1	→ 168
← Distance 30,500 m →						
1098	→ 3	Aquifer	30	10^{-5}	15	→ 336
Flow Regime	Case No.	Change From Base Case				
1	1	Upstream Potential in Stratum 3 Increased to 1403 m H ₂ O				
1	2	Stratum 2 Permeability Decreased to 10^{-9} cm/s				
2	3	Same as Case 1				
2	4	Same as Case 3 but with Stratum 2 Permeability Decreased to 10^{-9} cm/s				

FIGURE C-9 CROSS-SECTION DATA FOR EVALUATION OF FLOW REGIMES

repository strata and interbeds, this hydrostatic pressure imbalance dissipates as vertical flow occurs. The object of this set of calculations was to ascertain the magnitude of the vertical velocity and the distance at which the hydrostatic imbalance is dissipated as a function of assumed repository strata permeability.

The second set of cross-sectional runs (Cases 3 and 4) used the same basic geometry, but flow was assumed to enter and leave only the upper aquifer, with no recharge or discharge into the lower layers. Depending upon the permeability of the repository strata and interbeds, flow in the upper aquifer moved downward and spreads to the lower aquifer as well. The primary objective of these calculations was to ascertain the magnitude of the vertical velocity that might exist in the repository strata and interbeds.

The permeabilities used for the three strata were characteristic of the range for all of the strata types considered for HLW repositories. The values used for each case are illustrated in Figure C-9.

The calculated vertical velocities (Darcy or superficial, rather than interstitial) in the repository strata are shown in Figure C-10. The vertical velocities at the upstream end for the high-permeability cases (1 and 3) are larger by at least an order of magnitude than those of the low-permeability cases (2 and 4). But the hydrostatic pressure imbalance decreases more rapidly in the high-permeability cases, and the magnitudes of the vertical velocities decrease accordingly. Because of the more rapid dissipation of the hydrostatic imbalance in the high-permeability case, there is a region where the vertical velocities in the interbeds are actually higher for the low-permeability interbed case than for the high-permeability case, as shown in Figure C-10. In Cases 3 and 4, the vertical velocities reverse sign at the midpoint between the recharge and the discharge ends. The velocities are symmetrical around the zero velocity point and are higher near the discharge for Case 3 (higher permeability) than for Case 4. In both these cases, the calculated vertical velocity at the midpoint of the interval is less than 10^{-7} ft/day (Darcy velocity).

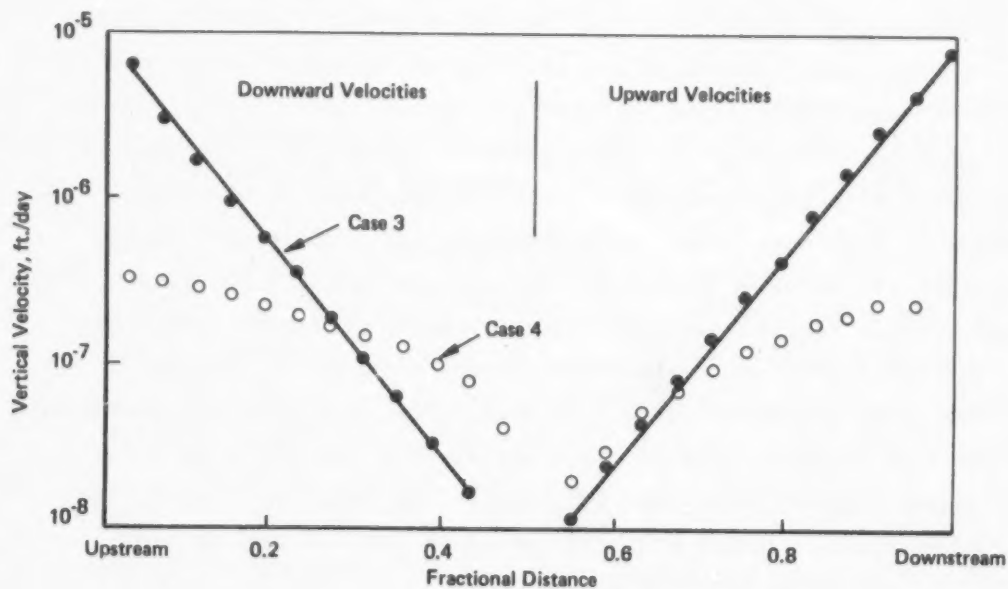
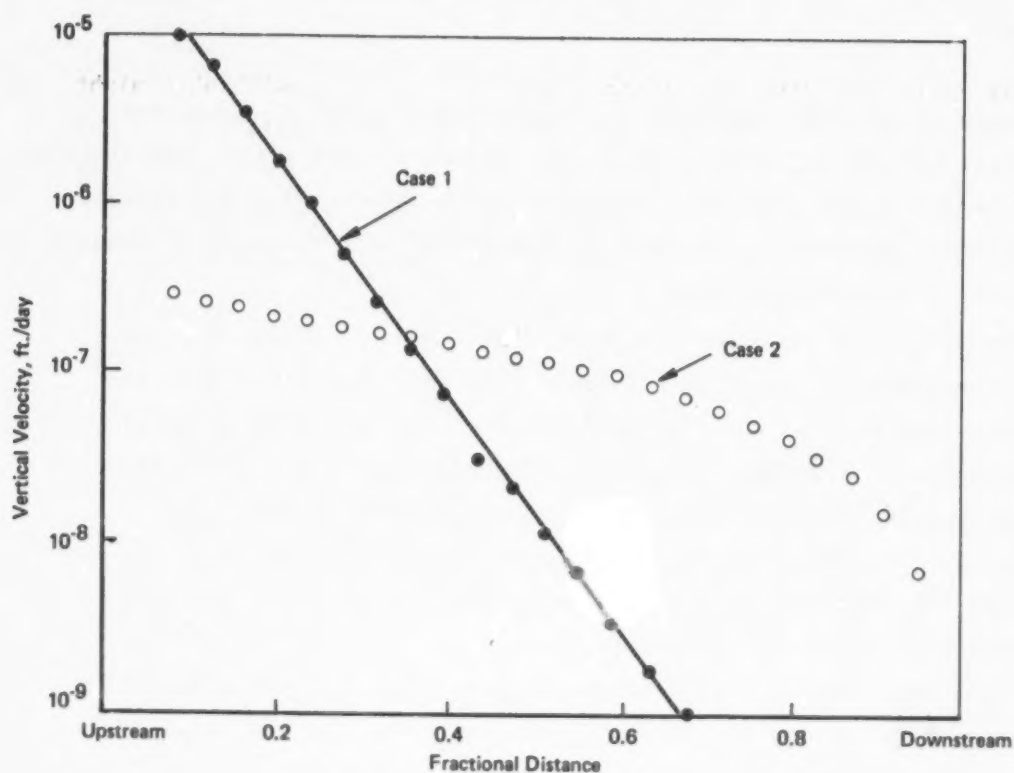


FIGURE C-10 CALCULATED INTERBED VELOCITIES FOR TWO FLOW REGIMES

The calculated vertical velocities can be compared with what might be termed "significant" flows. One definition of a significant vertical flow is one that would still allow a salt bed to exist over geologic time. A salt leach front (the point at which most of the soluble salt has dissolved) will move at only a fraction of the natural water flow. This leach-front velocity can be expressed approximately as:

$$v_l = \left(\frac{1}{1 - \phi_s} \right) \frac{v_p \rho_w X_s}{\rho_s}$$

where:

v = Darcy velocity of water,

v_l = leach front velocity,

ρ_w = water density,

ρ_s = solid salt density,

X_s = weight fraction of solid salt.

It should be noted that the salt porosity, ϕ_s , could probably be assumed to be zero without significant loss in generality.

For typical values in the above expression, the leach front could move about one-tenth as fast as the water velocity. At a natural Darcy water velocity of 10^{-7} ft/day, the salt leach front would move only about 30 cm (1 ft)/30,000 years. Leach rates higher by two orders of magnitude could probably be significant, but could probably not exist in or near existing salt beds. Nuclide migration at natural flow rates as low as 10^{-7} ft/day should not be significant, since it would require almost 3 million years to flow 30.5 m (100 ft).

Another type of water movement in the repository strata, although caused by the presence of the repository, might be termed "natural" flow. In considering potential repositories in certain existing salt beds, the presence of small pockets or volumes of saturated or super-saturated brine have been noted. These brine-saturated regions could migrate slowly toward the repository because of the temperature gra-

dient imposed by the radioactive wastes. Additional dissolution of salt on the high-temperature side of the gradient, along with recrystallization on the low-temperature side, could cause migration of the liquid. Although this migration could create substantial water contact with the waste, there would be no additional driving force for water migration away from the repository.

Similarly, water movement due to thermal stress has been observed for potential repositories in salt and shale. Again, these low-velocity flows would move toward the potential repository (heat source) rather than away from it.

Even if highly selective siting criteria are applied, there can be extremely slow natural migration of water through the repository strata. If this occurs, then the velocity of this flow would have to be so low, that, even if the nuclides were in soluble form and no sorption took place, the migration would not be significant.

A repository containing HLW creates a substantial heat source. The heat stress thus imposed may cause some fracturing of the repository and interbeds and open up a more permeable channel to an adjacent aquifer. Similarly, penetration by exploration or other types of holes, or failure of the repository entry shaft seal could create a permeable area above the repository. The importance of these areas of increased permeability in terms of nuclide migration is discussed in Section 4.4 of this report.

C-4.2 EVALUATION OF LEACH RATES

C-4.2.1 Introduction

Most waste forms for the radioactive nuclides are heterogeneous, slightly porous solids. Leaching of the HLW form may include several simultaneous, interdependent physical mechanisms. The movement of nuclides may occur inside crystal lattices, along crystal grain boundaries, through porous regions, along pore surfaces, and through interparticle voids and may involve other complex processes at the molecular level. This movement is generally accompanied by ion-exchange type chemical

reactions, vaporization and condensation, and dissolution and precipitation of leached isotopes. Typically, leaching rates are higher during initial contact with water, dropping significantly shortly thereafter.

Factors that affect leachability are porosity and fractures in the solid, temperature of the solid, chemical composition of the storage matrix material, solubility of the isotope being leached, and characteristics of the leaching solution such as pH, salinity, etc. Porosity and fractures determine the solid surface area directly in contact with the fluid. Solubility of the waste form depends upon temperature; a temperature increase of 25-100°C can increase the leach rate and the total amount of material dissolved by factors of 10-100. Most laboratory measurements have been made at temperatures at or near 25°C, although the waste material buried in a geologic formation will probably be at higher temperatures. In the leach model used for transport calculations, a correction has been made to account for this temperature effect. Because of the differences in solubility, leachabilities of various isotopes in the same waste form material are not the same. This is especially true of the actinides, as will be discussed in Section C-4.4.

Principally because of differences in experimental techniques, available leaching data are not always in agreement. The International Atomic Energy Agency (IAEA) adopted a standard procedure for measuring leachability of solids and expressing leach rates. Most of the published leach rate data are expressed in grams of solid leached/cm² of surface area/day. IAEA has recommended plotting the volume of solid leached as cm³/(cm²)(day), or cm/day units, or alternatively as the fraction of initial radioactivity leached vs. the square root of time. A good summary of test methods and leach data has been published by Mendel. ⁽³⁰⁾

C-4.2.2 Empirical Leach Models

The most commonly used empirical model for quantity leached is expressed as follows:

$$Q = at^{1/2} + bt \quad (10)$$

where

Q = cumulative quantity leached,

t = time, and

a, b = empirical constants.

The fit of this model to much of the experimental data shows that the variation of Q is proportional to $t^{1/2}$ for short time periods and linear for longer times. The above expression for cumulative quantity leached would indicate a constant leach rate is approached at longer times.

Much of the data show this behavior after leach periods of only 2500 days or less.

Leach data obtained at Oak Ridge National Laboratories have been fitted to the equation:

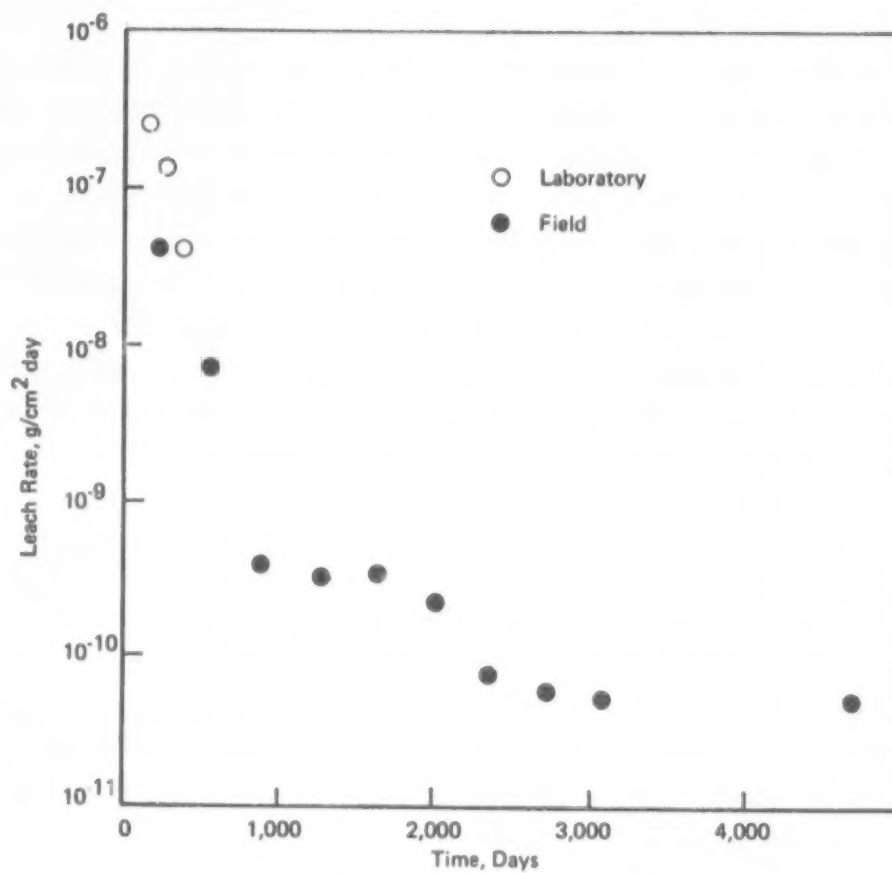
$$Q = at^a \quad (11)$$

A similar expression was used by Zagar and Schillmoeller for their data for leaching from glass.⁽³¹⁾ As pointed out in this work, when a is 0.5, the constant " a " can be related to a diffusivity in the porous solid.

Investigators at Atlantic Richfield Hanford Company have fitted experimental data with a polynomial equation, attributing each term in the equation to the leaching effect from a separate mechanism.⁽³²⁾

These empirical leach models fit much of the laboratory leach data. An important aspect of the problem, however, is how consistent these laboratory data fit in situ waste leach rates. An in situ test has been conducted at the Chalk River site in Canada over the last 15 years.⁽³³⁾ Data from this work appear to be consistent with the laboratory leaching data on the same glass form, as shown in Figure C-11. A constant leach rate was achieved in the in situ test after 6-8 years in place. These data show an extremely low leach rate, stabilizing at about $5 \times 10^{-11} \text{ g/cm}^2/\text{day}$.

Leach rates at actual waste repositories would be even more complex, because of the substantial temperature effect from the radioactive decay. Based upon the expected temperature decline and changes in surface area as a function of time, an empirical model for the in situ waste form was derived in Task B, where:



Adapted from: Merritt, W.F. High Level Waste Glass: Field Leach Test.
Nuclear Tech., 32, January 1977.

FIGURE C-11 COMPARISON OF LABORATORY AND IN-SITU LEACH DATA

- (1) For glass forms (standard and devitrified) the fraction leached is proportional to the logarithm of time after water contact has occurred.
- (2) For calcine forms, the fraction leached increases linearly and relatively rapidly after water contact has occurred. No benefit in terms of a decrease in leach rate due to decline in decay heat was assumed.

In addition, many of the radioactive nuclides in the actinide chains have extremely limited solubility. Nuclides in other than the actinide chain can also exhibit solubility limitations. Under oxidation-reduction conditions existing in the natural groundwater, several nuclides might exist in a relatively insoluble form. Such nuclides include Tc-99 and Np-237, which under some circumstances would not be strongly adsorbed. In the actinide chains, U, Pu, Th, Am, and others might also be in a very insoluble form. Consequently, even though the waste form leaches, the nuclide may be present in an immobile, insoluble form. The importance of this solubility limitation in the migration of the actinide series has been examined and the results are discussed in Section C-4.4.

C-4.2.3 Theoretical Leach Models

Virtually all of the theoretical modeling efforts are solutions to one-dimensional differential equations in linear, cylindrical, or spherical coordinates.⁽³⁴⁾ These equations describe diffusion in a slightly porous solid with decay and formation of radioactive isotopes; in these equations, dissolution of the solid is important. The resulting solutions are complex because of the form of the equation and the initial and boundary conditions.

Solutions to the one-dimensional differential equations for theoretical leach rate are consistent with the empirical models.^(34, 35) These solutions show an initial leach rate that decreases rapidly with time and later becomes constant. The advantage of the theoretical solutions is that the constants can be interpreted as physical quantities, such as the diffusion coefficient.

The approach in modeling geosphere transport has been to use the leach rates defined in Task B. Two computer runs investigating different assumptions about waste leaching were made. One run was consistent with Figure B-42 of the Task B report for the amount of glass leached as a function of time. That curve represented a leach rate decreasing with time only because of the changing surface-to-mass ratio. No separate correction for the temperature effect was made. The second run assumed that the initial leach rate had been increased by a factor of four by the repository temperature increase. In this run, leach rates beyond 4000 years were less than those shown in Figure B-42 to remove some of the conservatism. The fraction of the waste leached as a function of time for both runs is plotted in Figure C-12.

C-4.3 GEOSPHERE TRANSPORT THEORY ASSESSMENT

As previously mentioned, the objective of this study has been to investigate non-site-specific parameters important in mitigating radionuclide transport in the geosphere. Formulation of the cases to be examined with the model is strongly dependent upon this non-site specificity.

A distinction should be made regarding the range of values for individual parameters (permeability, porosity, adsorption, etc.) needed to describe the classes of generic sites and the uncertainty in the identity of these same parameters once a specific site is selected. In the former case, the parameters might need to span the range of salt, basalt, granite, and other types of formations. For the specific site case, a different range of parameters may be needed to describe the uncertainty remaining subsequent to a measurement program. Modeling by the petroleum industry supports this conclusion.

These petroleum models solve a broad range of equations for simultaneous three-phase flow, energy considerations in enhanced recovery mechanisms, multicomponent oil phases, and other complexities. In addition, a number of laboratory measurements, as well as in situ tests, are made in

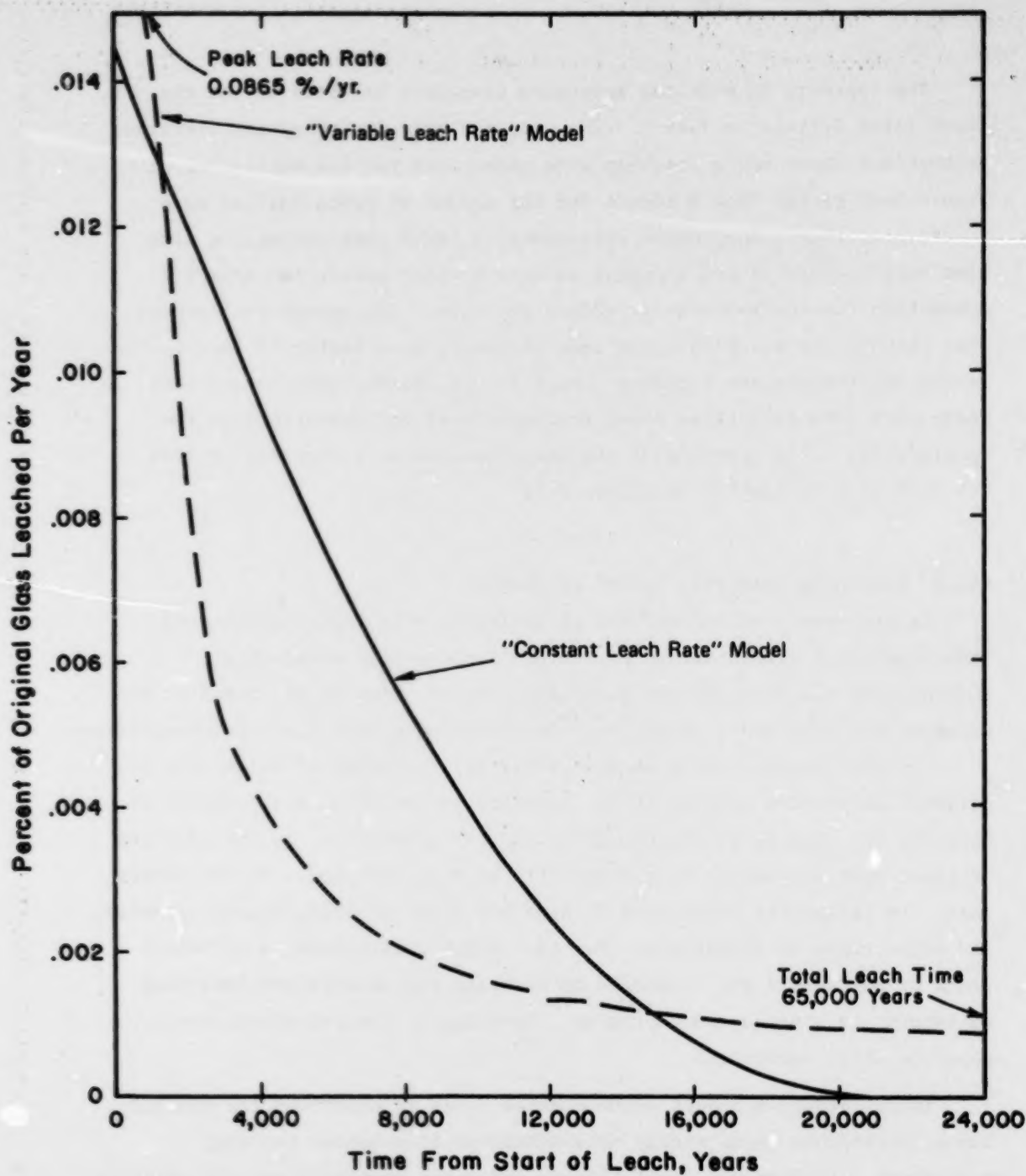


FIGURE C-12 COMPARISON OF LEACH RATES FOR TWO LEACH MODELS

order to provide background data for these models. Significantly few, if any, purely predictive models have been developed. On the contrary, the models are first required to fit existing history concerning a specific reservoir. Use of such models generally involves four steps:

- (1) Compile data concerning both the geologic description of the reservoir and the fluid properties.
- (2) Utilize these data in the model, and attempt to match model output with the reservoir performance history. (This would include production of oil, gas, and water at each well, along with measured pressures with the wells either producing or shut-in).
- (3) Adjust these data as necessary and to the extent consistent with the geologic description in order to obtain the best match to the existing reservoir history.
- (4) Predict future reservoir performance.

Petroleum industry practice is cited to emphasize that uncertainty in both field and laboratory measurements is well recognized. This does not imply that the measurement itself is inaccurate, but rather that it is impossible to use these point-value measurements (whether laboratory or in situ) as necessarily representative of a geologic transport response over an area of the size of a repository. Mathematical predictions for non-site-specific areas or for geologic strata for which the available data are limited must have considerable uncertainty. Sensitivity calculations are, therefore, required for a range of the parameters important to the transport process.

The complexity of a model suitable for such non-site-specific calculations is an important consideration. There is a potential for error in using too sophisticated a model, thereby perhaps hiding some of the uncertainty that actually exists, or in using too simple a model that may not include potentially important parameters. The results of a rather complete review of models that could have application in radioactive nuclide transport will soon be published.⁽³⁶⁾

C-4.3.1 Hydrology - Radionuclide Transport Model Review

An extensive review of hydrologic models has been completed by the Holcomb Research Institute with the support of EPA.⁽³⁶⁾ Results from this

survey are not yet available in published form, but summaries and conclusions from that review were discussed with members of the review team.

This review was quite comprehensive and conducted on a worldwide basis. Questionnaires were mailed to groups known to be working in the field of hydrology soliciting information concerning the types of models developed and available. Later, workshop sessions were held at three locations in the United States in order to obtain additional information regarding actual applications and limitations of existing models. Those models useful for geosphere radioactive nuclide transport are discussed here.

Any model to assess nuclide transport has two essential parts, (1) the aquifer flow; and (2) the transport, dispersion, sorption (including precipitation), and radioactive decay of nuclides in this flow. These two parts can generally be uncoupled and solved independently. Two critical factors must be considered, however, before such uncoupling can be permitted. The first critical factor to be considered is the concentration of the radionuclides. If the geologic strata represent a viable repository, nuclide concentrations will have to be so very small that they will not affect the density or viscosity of the water.

The second factor is the amount of heat that will be generated as a result of the energy released by decay of the radioactive nuclides. The density and/or water viscosity could be affected by increases in temperature and could initiate natural convection. Since the leached nuclides should be at trace concentrations in the water, however, the heat generation within the flowing water should be negligible, thus removing this possible barrier to uncoupling. This latter effect would not normally be negligible, but instead of coupling the water flow and nuclide transport, it would instead couple flow with energy balance.

The following comments are a summary of the review of available models for geosphere nuclide migrations:

- (1) A number of hydrologic flow models are available that consider a rather complete range of spatial dimensions, inhomogeneities, anisotropy, and other factors.

- (2) A number of constituent transport models are also available that are less general than the flow models but are quite usable for consideration of trace-level constituents that undergo simple radioactive decay.
- (3) Only two models appear to include both the decay and growth exhibited during parent-daughter radioactive chain interactions.* These two models are quite different in approach.

One, developed at Battelle Northwest Laboratories, represents an analytical solution to the set of one-dimensional partial differential equations describing dispersion, sorption, convection and radioactive growth, as well as decay, of each nuclide.⁽³⁷⁾ Since it is based upon an analytical solution, this model is limited to the following:

- (a) Dissolution of the waste form at a constant rate (radioactive decay of the waste form is included).
- (b) Water flow at a constant velocity.
- (c) Trace levels of the nuclides are present, so non-interactive equilibrium sorption can be assumed.
- (d) Hydrodynamic dispersion is treated as constant and one-dimensional in the axial flow direction.

* Other constituent transport models have been used by various authors to analyze radionuclide migration; however, these do not appear to have included the interaction of complete parent-daughter chain reactions (e.g., see (35) and (38)). It should be noted that most models that include nuclide source and decay terms could be used for full parent-daughter chains by a successive application of the nuclide balance through the entire chain. Examples of models that have used this concept include (39).

When restricted to constant flow velocity, the flow model can be uncoupled from the transport code, and the constant velocity would be reduced to a simple steady-state flow approximation. For situations where the water velocity is not constant, however, the model could be used by assuming a representative, composite average flow.

The other model, which includes complete geosphere transport and decay chains, is one recently developed by INTERA Environmental Consultants for Sandia Corporation on behalf of the U.S. Nuclear Regulatory Commission (NRC).⁽⁴⁰⁾ A brief summary of this model appears as Appendix C-II of this report.

This model represents a numerical finite-difference solution to the set of partial differential equations describing geosphere transport. It uses second-order, correct spatial finite-difference forms to eliminate numerical dispersion, although in doing so, it places a limitation on the size of the grid block and the time step that can be used. The model, similar to the one previously described, allows for hydrodynamic dispersion, sorption, convection, and the nuclide chain radioactive decay and growth. This model offers considerably more flexibility, in that it is multidimensional and handles inhomogeneous geologic strata or groups of strata. Moreover, the model couples the nuclide transport code with a complete multidimensional water flow model. This model still assumes trace levels of radioactive constituents, so the sorption properties can be treated as if in equilibrium and non-interacting. Furthermore, the model can be used to examine thermal convection effects or a constituent that affects water density, since it includes a total-energy balance and a major-constituent balance.

For the present application involving geosphere transport, most of the assessment was made for a one-dimensional geosphere pathway. As a consequence, either the analytical transport model or the numerical transport model described above could have been used for the calculations. Actually, both models have been used for this study: the analytical model

for simple sensitivity calculations, and the more flexible numerical model for the actual inhomogeneous migration path from the repository to the aquifer. Two scenarios have been calculated with the numerical model to provide concentrations to the biosphere pathway models for dose-to-man calculations in Section C-5.

Most, if not all, of the models included in this review and discussed above use a "continuum" approach (transport through a non-fractured porous medium). Much of the transport in the immediate vicinity of a potential repository may be in multiple-channel fractured media, however. Furthermore, many potential repository rock types can exhibit natural permeability and/or flow characteristics based upon a fracture system. Nevertheless, many of the natural fracture systems containing petroleum reserves have been modeled using a continuum approach. Gas production in a number of places has also been modeled with a continuum approach through a dual porosity system, with the fracture system characterized by significant permeability despite a low porosity. Such fractures communicate by a diffusional mechanism (capillarity, heat conduction, or component diffusion) with a rock matrix that has porosity, but essentially no flow permeability.

The Office of Waste Isolation recently sponsored a workshop on "Movement of Fluids in Largely Impermeable Rocks"⁽⁴¹⁾. The discussion focused on water movement, largely through fracture systems, in low-permeability rocks. One of the tasks in this workshop was to investigate the role of numerical modeling in evaluating transport of chemical species and heat in such an environment. The participants concluded that longer-term effects (1000-1 million years) could perhaps be simulated by continuum models. Shorter-term effects, however, might require models that include the actual fracture networks.

The mathematical description of laminar water flow in fractures is not fundamentally different from that of continuum flow through a porous rock. That is, in both cases the flow velocity is proportional to the hydraulic gradient (Darcy's Law). However, the permeability term in Darcy's Law then becomes proportional to a power (usually the square) of the fracture spacing. Thus, the question of fracture model versus the continuum model for water flow is not a difference in model type but rather a difference in how to characterize, from either deterministic or stochastic considerations, the fracture spacing, width, and geometry.

Mathematical description of the nuclide transport would be described essentially in the same way, whether the flow were in fractures or in a porous medium. If the matrix blocks are not porous, only transport in the fracture system would be calculated. Nuclide adsorption on the fracture face would need to account for a different ratio of surface area to volume than for porous systems. In the case where the matrix blocks include effective porosity, a dual porosity approach would be necessary. In this case, diffusion of nuclides into the matrix blocks slows the net transport rate of the nuclides in the fracture. In the present study, a continuum model has been used to calculate the water and nuclide transport in both porous and fractured systems. The effective porosity of the fracture systems has been assumed to be small, and no diffusion into matrix blocks has been included. This is a conservative approach, in that it allows the nuclides to move slightly faster than they actually would.

In the modeling used for this analysis of geosphere migration for fracture systems (whether naturally present or created subsequent to repository installation) the fractures are assumed to offer the permeability for transport but to have quite low porosity. Consequently, the rate of nuclide migration in these fractured regions can be significantly higher because of a higher interstitial or transport velocity. Another factor that increases the transport rate of the radionuclides is the reduced adsorption that could take place on the fracture surface.

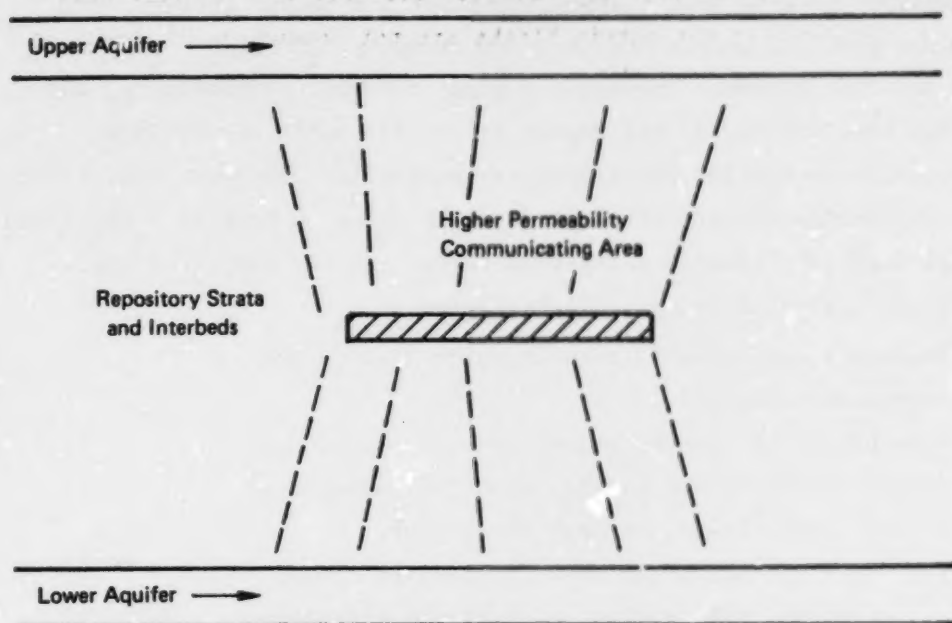
C-4.4 MIGRATION OF RADIONUCLIDES IN THE GEOSPHERE

C-4.4.1 Introduction

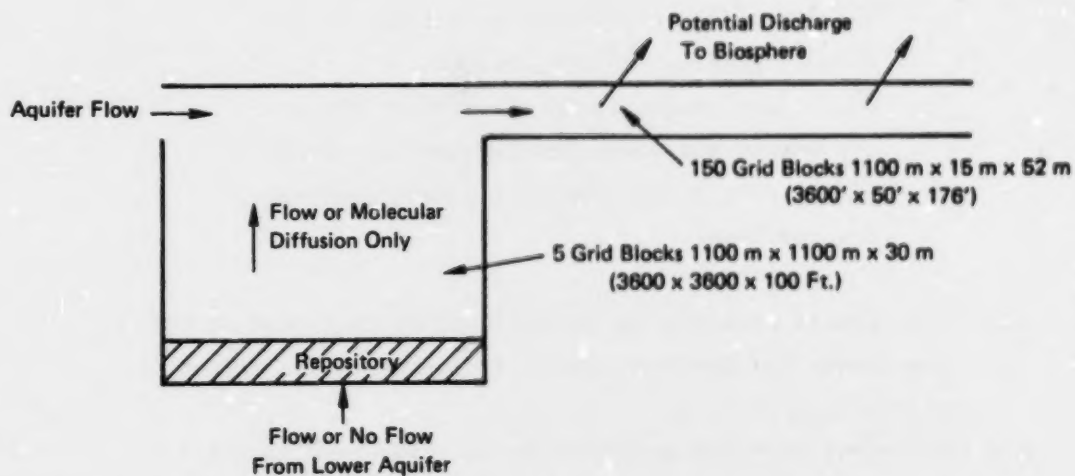
C-4.4.1.1 Modeling Concepts

In Section C-3.4, conceptual models were developed that are representative, in a generic sense, of geologic media that could serve for HLW storage. These potential repository strata include bedded salt, two types of shale, basalt, and an intrusive salt dome. For nuclide migration modeling, the essential aspects are the presence of adjacent aquifers above and/or below the repository strata and the communication that exists or could develop between the repository formation and these aquifers. The interbedded layers between the repository strata and the aquifer(s) also are an important part of this flow regime.

Each of the conceptual geologic models shown in Section C-3.4 is amenable to a similar type of calculational model for geosphere transport. This is illustrated in Figure C-13 as one-dimensional migration both in



(a) Schematic Vertical Section of Model Area



(b) Model Dimensions

FIGURE C-13 NUMERICAL REPRESENTATION OF GEOLOGIC MODELS

the repository and interbedded strata, and in an aquifer. Although the geometry modeled appears more characteristic of the bedded geology, even the intrusive geologic models are not significantly different. The pathway still involves movement through a region of higher permeability, which movement has been induced in the repository strata, with the nuclides finally reaching an aquifer for horizontal migration. For modeling, there is little difference between the five geologic types as long as a continuum approach is used to represent a fractured area. In the numerical transport model the following factors can be included:

- (1) Extremely low-velocity natural water flow in the repository strata.
- (2) Possibility of communication, because of thermal stress or disruptive events, from the repository to the upper, lower, or both aquifer(s).
 - (a) This communication could consist of liquid contact, but no flow, with liquid diffusion of nuclides to the aquifer.
 - (b) The communication could allow a small flow from the aquifer down into the repository and returning to the aquifer because of the natural gradient in the aquifer.
 - (c) The communication could allow flow from one aquifer to the other through the repository if there is a hydrostatic imbalance between the aquifers.
- (3) Sorption of nuclides can be included or neglected in the repository and interbed layers independent of sorption in the aquifer.
- (4) Discharge from the geosphere to the biosphere can be calculated at many points along the pathway within the aquifer.

In the analyses, properties of the communication area, as well as those of the aquifer, were varied in order to show the importance of these effects in geosphere transport.

C-4.4.1.2 Sensitivity Analysis Modeling

Both the analytical and the numerical models were used to perform sensitivity analyses. Two types of parameters were investigated: those that affect the transport mode, and those that affect the leaching characteristics. The parameters tested and the model used for each test were as follows:

<u>Transport Mode</u>	<u>Parameter</u>	<u>Model</u>
	Repository Bed Thickness	Analytical
	Permeability	Analytical
	Flow Gradient	Analytical
	Adsorption	Numerical
	Dispersivity	Analytical
	Porosity	Analytical
<u>Leach Characteristics</u>		
	Leach Time	Analytical
	Leach Rate	Analytical
		Numerical
	Nuclide Stability	Numerical
	Containment	Numerical
	Distance to Biosphere	Analytical

The ranges of values of the parameters that affect the transport mode are shown in Table C-7. These values have been taken from Tables C-1 and C-2 for low- to moderate-permeability sandstone formations. Assuming that site selection criteria would eliminate the upper end of the permeability range, the range included in Table C-7 seems reasonable. A similar tabular listing of ranges for the geosphere transport parameters was subsequently determined by a panel of personnel from the U.S. Geological Survey, the EPA, and INTERA. This list is summarized in Appendix C-III. The range of parameters is similar in most respects to those in Table C-7, although some of the parameters cover a wider range.

C-4.4.1.3 Nuclide Concentration Modeling

The purpose of this modeling was to calculate nuclide concentrations for use as input for assessment of the radiation dose to man in Section C-5. Two sets of calculations were made. Scenario 1 corresponds to a liquid diffusional pathway in the repository and interbed strata. It also assumes that the permeability of the repository and interbedded

TABLE C-7

GEOSPHERE MIGRATION PARAMETERS AFFECTING TRANSPORT

<u>Variable</u>	<u>Parameter Range</u>	
	<u>Aquifer</u>	<u>Communication Area</u>
Thickness, m (ft)	15-45 (50-150)	150-450 (500-1500)
Permeability*, cm/s (ft/day)	10^{-6} - 10^{-4} (2.8×10^{-3} - 2.8×10^{-1})	10^{-9} - 10^{-7} † (2.8×10^{-6} - 2.8×10^{-4})
Flow gradient, m/m (ft/ft)	0.01-0.1	0-1.0
Adsorption, K_d (ml/g)	(0.1-1.0)x(normal)**	(0-2.0)x(normal)**
Dispersion, m (ft)	6-60 (20-200)	60 (200)
Porosity %	15	1-10

*Vertical permeabilities were one-tenth that listed.

†Fracture permeabilities in communication area varied from 10^{-7} to 10^{-5} cm/s, and porosity values (including fracturing) were always taken as 0.01.

**See Table C-3.

strata has been increased by fracturing to 100 times higher than normal. The scenario also uses the most permeable aquifer and highest flow gradient to develop the greatest case flow condition in the aquifer. The 100-fold increase in permeability for an interbed region permeability (conductivity) of 10^{-7} cm/s assumes that a fracture of 0.1 mm every 3 m has been created throughout the region above the repository. The leach rate has been chosen as that for a standard glass waste form, with no solubility limitation. The throwaway cycle waste started leaching at a waste age of 200 years. "Normal" adsorption (defined by the retardation factors) was used.

Scenario 2 utilizes the same aquifer flow conditions, but is based upon what might be termed a "worst probable" case. It consists of connecting a lower aquifer to an upper aquifer by flow through the repository.

Analysis of these two scenarios identified the nuclides most important in the migration pathway. These nuclides tend to have similar characteristics, such as:

- low adsorption,
- long half-life,
- they, or their parent components, are present in large initial amounts.

Migration of most important nuclides, such as: Np-237, I-129, Sr-90, Ra-226, Cs-137, Tc-99 (when no adsorption is assumed) and some of the plutonium and uranium nuclides, is discussed in Sections 4.4.3.2 and 4.4.3.3.

C-4.4.2 Results Using Analytical Model

With the analytical model, the sensitivity analysis examined primarily the following input variables:

- leach time,
- hydrodynamic dispersivity (a multiplier that is applied to velocity to produce a total dispersion coefficient),
- water migration rates (in turn a function of permeability, porosity, and hydraulic gradient),

- repository and interbed thickness, and
- distance to discharge to the biosphere.

Two output values are important: (1) the initial arrival time at a given distance from the repository, and (2) the dilution in nuclide concentration that occurs during migration. The arrival time is most affected by water migration rate and nuclide adsorptivity. Dilution is affected by both these parameters and is also sensitive to leach time and hydrodynamic dispersivity.

The partial differential equation describing nuclide transport and the analytical solution for a single decaying adsorbing nuclide are shown in Appendix C-II. When decay is not significant over the travel time of the nuclide, the analytical solution can be reduced to an extremely simple form for evaluation of arrival and dilution time:

Initial Arrival Time, τ

$$\tau = \frac{LR_d}{v_w} \quad (12)$$

where

- L = distance to biosphere discharge,
- R_d = adsorption retardation factor, and
- v_w = the interstitial water velocity,

Dilution

$$C = C_o \operatorname{erf} \left[\frac{v_w}{R_d} \frac{T_L}{L} / 4\sqrt{\alpha/L} \right] \quad (13)$$

where

- C = the maximum concentration at L ,
- C_o = the boundary concentration at the repository,
- T_L = the leach time,
- α = the hydrodynamic dispersivity, and
- $\operatorname{erf}(x)$ = the standard tabulated error function.

The approximate dilution expression above can be derived from the analytical solution given in Appendix C-II by noting that peak concentration will occur at the midpoint of the traveling body of material. These expressions can be used to obtain a sensitivity analysis of several variables.

In all calculations, the repository geometry was assumed to be constant and consistent with the repository handling wastes from reactors charged with 50,000 MTHM. For heat dissipation, the site was assumed to have a loading of 150 kW/acre. In the calculations for this study, the waste was assumed to occupy a 300-acre site.

Table C-8 summarizes the I-129 arrival times and relative concentrations at specified points. Iodine adsorption is zero. The two water velocities correspond to the highest levels listed in Table C-7 for conditions in the aquifer, i.e., a hydraulic permeability of 10^{-4} cm/sec, a gradient of 0.1 m/m, and a porosity of 15%; and to the lowest values, which are 10^{-6} cm/sec, 0.01 m/m, and 15% respectively. The travel times to a point 1.6 km (1 mile) distant from the repository range from 77 years up to almost 77,000 years.

The masses of the significant actinide and fission product nuclides initially present in the repository are listed in Table C-9. The mass of I-129 present for the throwaway cycle is 1.16×10^7 g. The flow for a 15 m (50 ft)-thick aquifer over an 1100 m (3600 ft)-wide repository would be approximately 144,000 liters/day and 144 liters/day for the high- and low-flow cases, respectively. Consequently, if all of the I-129 is assumed to be leached at the high flow rate in one year, the average concentration would be 0.22 g/liter and if leached in 100 years, the average concentration would be 0.0022 g/liter. This results from the initial concentration being inversely proportional to the product of water velocity multiplied by leach time. The effect of leach time on initial nuclide concentration is illustrated schematically in Figure C-14. Thus, for a leach time of one year and a dispersivity of 3 m, the maximum concentration of I-129 at 8 km (5 miles) from the repository would be 0.0084 g/liter. On the other hand, for the longer leach time of 100 years, even with no dispersion in the geosphere transport path, the maximum concentration would be 0.0022 g/liter. This is simply an indication that, for a soluble, non-sorbed, long-lived nuclide such as I-129, the important reduction in nuclide concentration is due to the leaching rate limitation. In case of the larger dispersivity of 30 m, the concentrations of I-129 at 8 km (5 miles) distant from the repository are 0.0026 g/liter and 0.0019

TABLE C-8

I-129 ARRIVAL TIMES AND RELATIVE CONCENTRATIONS

Interstitial Water velocity (cm/day)	Leach Time (yr)	Dispersivity (m)	Distance-1.6 km (1 mile)		Distance-8 km (5 miles)	
			Arrival Time (yr)	Relative Concentration*	Arrival Time (yr)	Relative Concentration*
5.75	1	3		0.085		0.038
		30		0.027		0.012
	100	3	77	1.000	383	1.000
		30		0.996		0.860
.00575	1	3		8.5×10^{-5}		3.8×10^{-5}
		30		2.7×10^{-5}		1.2×10^{-5}
	100	3		0.0085		0.0038
		30	76,674	0.0027	383,368	0.0012
	1000	3		0.085		0.038
		30		0.027		0.012

*Concentration at specific distance relative to concentration at aquifer entry.

TABLE C-9
RADIONUCLIDE AMOUNTS FOR ORIGEN OUTPUT

(a) Actinides and daughters

Nuclide	Half-Life (Yr)	Retardation Factor, R_d	Initial Mass in Place*		
			(g)	(g)	(g)
			(Case 1) Throwaway	(Case 2) UO ₂ Recycle	(case 3) Mixed Oxide
Cm-246	4700	3300.	1.48×10^3	1.48×10^3	3.18×10^5
Am-242m	152	10,000	2.26×10^1	2.26×10^1	4.95×10^2
Pu-242	379,000	10,000	2.24×10^7	1.12×10^5	2.57×10^6
U-238	4.51×10^9	14,300	4.72×10^{10}	2.36×10^8	2.26×10^8
Pu-238	86	10,000	6.50×10^6	2.14×10^5	3.87×10^6
U-234	247,000	14,300	9.15×10^6	5.90×10^4	†
Th-230	80,000	50,000	2.91×10^2	5.85×10^1	2.19×10^{-7}
Ra-226	1600	500	1.48×10^{-2}	5.50×10^{-3}	1.97×10^{-11}
Cm-245	9300	3300	1.25×10^4	1.25×10^4	4.46×10^6
Pu-241	13.2	10,000	3.91×10^7	1.96×10^5	1.67×10^6
Am-241	458	10,000	2.53×10^7	3.28×10^6	5.25×10^7
Np-237	2.14×10^6	100	2.35×10^7	2.33×10^7	6.90×10^6
U-233	162,000	14,300	3.60×10^2	7.20×10^1	2.59×10^2
Th-229	7340	50,000	2.10×10^{-2}	9.60×10^{-3}	1.35×10^{-3}
Cm-247	1.6×10^7	3300	1.99×10^1	1.99×10^1	4.20×10^3
Am-243	7370	10,000	4.46×10^6	4.46×10^6	1.27×10^8
Cm-243	32	10,000	3.60×10^3	3.60×10^3	3.62×10^4
Pu-239	24,400	10,000	2.70×10^8	1.36×10^6	3.76×10^6
U-235	7.1×10^8	14,300	4.02×10^8	2.01×10^6	9.50×10^7
Cm-248	470,000	3300	1.41×10^0	1.41×10^0	2.44×10^2
Pu-244	8.0×10^7	10,000	2.56×10^{-5}	2.38×10^{-5}	4.12×10^{-3}
Pu-240	6580	10,000	1.11×10^8	9.10×10^5	1.68×10^7
U-236	2.39×10^7	14,300	2.05×10^8	1.03×10^6	1.54×10^5
Th-232	1.4×10^{10}	50,000	6.70×10^1	1.17×10^1	1.47×10^0

*Based on 50,000 MTHM charged to the reactors, and a 10-yr cooloff period.
†Value not listed in ORIGEN.

TABLE C-9
(Continued)

(b) Fission Products

<u>Nuclide</u>	<u>Half-Life (Yrs)</u>	<u>Retardation Factor, R_d</u>	<u>Initial Mass in Place*</u> (g)		
			<u>(Case 1) Throwaway</u>	<u>(Case 2) UO₂ Recycle</u>	<u>(Case 3) Mixed Oxide</u>
H-3	12.3	1	2.15×10^3	1.72×10^2	2.27×10^2
C-14	5730	10	1.71×10^2	1.71×10^2	2.17×10^1
Kr-85	10.8	1	7.65×10^5	0	0
Sr-90	27.7	100	2.12×10^7	2.12×10^7	1.23×10^7
Tc-99	2.1×10^5	{ 1 3300	4.20×10^7	---	---
I-129	1.7×10^7		1.16×10^7	1.46×10^4	1.86×10^4
Cs-134	2.05	1000	3.53×10^5	3.53×10^5	1.55×10^5
Cs-137	30.0	1000	4.96×10^7	4.96×10^7	5.10×10^7
Sm-151	87	2500	2.14×10^6	2.14×10^6	3.62×10^6
Eu-154	16	2500	1.79×10^6	1.79×10^6	1.24×10^6

*Based on 50,000 MTHM charged to the reactors, and a 10-yr. cooloff period.

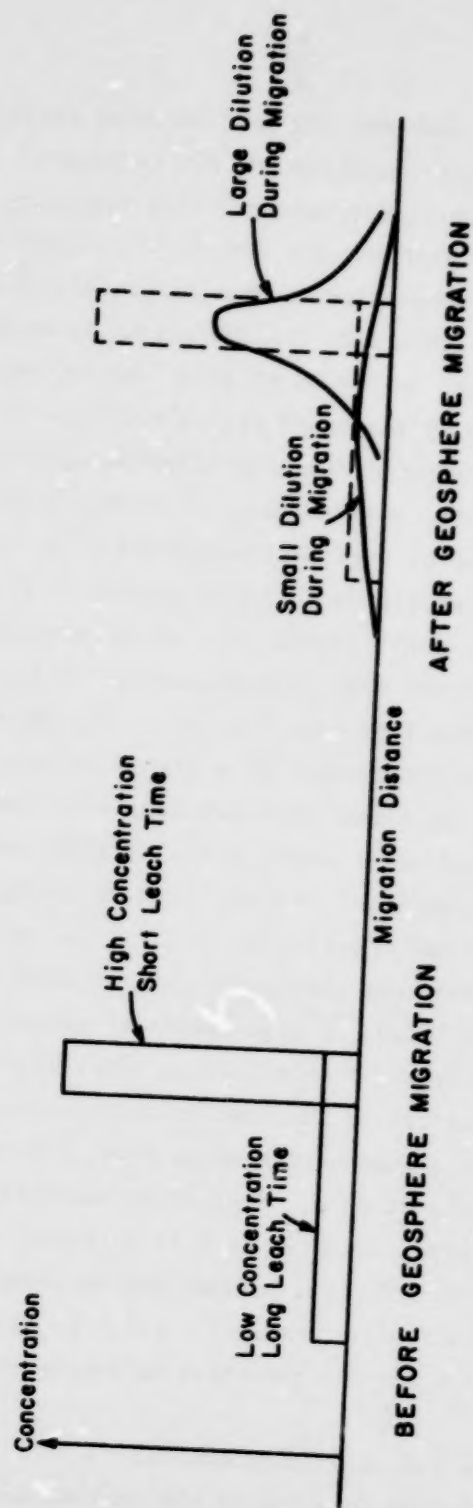


FIGURE C-14 EFFECT OF LEACH TIME AND MIGRATION ON NUCLIDE CONCENTRATION

g/liter, respectively, for the 1-year and 100-year leaching periods. Even in this case, the reduction in concentration due to reduced leach rate has been larger than that due to dispersion during migration.

At lower rates of nuclide migration, the reduction in concentration in the geosphere pathway due to dispersion becomes relatively more important. For the low rate illustrated in Table C-8, the end result is somewhat surprising. It indicates that the increased dilution due to longer leach times is exactly compensated for by decreased dilution during the geosphere pathway migration. This can be deduced from the dilution equation, since the boundary concentration leaving the repository (for this idealized "constant leach rate" case) would be inversely proportional to the water velocity. The reduction in concentration during migration is proportional to the water velocity, which for small values is a linear proportion. Thus, for small water velocities (or high retardation factors), there will be an interdependence between time and flow rate. The above effect can be noted in Table C-8, where the result for a high flow rate and one-year leach period is identical (in final concentration) to a flow rate lower by a factor of one thousand and a leach period of 1000 years.

Table C-8 shows that for long-lived nuclides such as I-129, half-life has little effect. (I-129 has a half-life of 1.7×10^7 years.) If Np-237 were not strongly adsorbed, the result for this nuclide would be virtually identical to that for I-129. A high degree of adsorption ($R_d = 100$) reduces the concentration of this isotope after migration through a distance of 8 km (5 miles) in the aquifer to only about 1.2% of that leaving the repository, assuming high water flow, 100-year leaching time, and a dispersivity of 30 m. Neglecting adsorption, (i.e., assuming $R_d = 1$), the concentration of Sr-90 at 8 km (5 miles) is reduced to 3.5×10^{-5} of the initial concentration leaving the repository. Application of the probable adsorption value for Sr-90, which is about equal to that for Np-237, would produce a further reduction in concentration at 8 km.

These results lead to the following conclusions:

- (1) If the leach time is more than 25% of the nuclide migration time, hydrodynamic dispersion will be relatively unimportant.

Reduction of concentration during geosphere transport will result primarily from radioactive decay.

- (2) Retardation of migration by adsorption in the geosphere is important in reducing concentrations of nuclides, as well as in delaying nuclide arrival time at specific points.

This effect on concentration is due to the increased time permitted for dispersion to occur.

The effect of total distance--the thickness of the repository itself and interbeds--between the repository and an aquifer is also important. The type of analysis examined above can be applied to this variable. Although the vertical path length in the repository-interbed section would be considerably shorter than the horizontal distances investigated above, the vertical permeability would also be much lower, and thus there is a compensating effect. Therefore, a 150 m (500 ft) thickness for repository and interbed strata will be similar in its effect on nuclide travel time to the distance between 0 and 1.6 km (1 mile) in the aquifer, as illustrated in Table C-8.

Another variable concerns the effectiveness of the containment, i.e., when leaching starts. For most radionuclides, later leach initiation is advantageous, because concentrations have then decreased because of radioactive decay. A few daughter nuclides, e.g., radium-226, however, exhibit increases in concentration for up to 100,000 years and subsequently start decreasing. In such circumstances, the early release could reduce concentrations at a later time. Unless Ra-226 became the most critical nuclide from the standpoint of dose, early release would be disadvantageous. The concentrations of most, if not all, of the other critical nuclides decrease with time and their earlier release would be highly disadvantageous.

No detailed set of calculations has been made for the fuel cycles other than throwaway (Case 1, Task A Report). In general, wastes from the other cycles have less of the critical nuclides, e.g. I-129, Ra-226, Np-237 and plutonium isotopes. A few nuclides are increased, e.g. Am-243. If such nuclides as Am-243 contribute a

significant amount to the potential dose rate, these cycles should be investigated.

The calculations discussed above used the analytical model previously described. The remaining calculations were made with the numerical nuclide transport model.

C-4.4.3 Results Using Numerical Model

C-4.4.3.1 Parameters for Scenarios 1 and 2

The waste form placed into the repository was assumed to be from the throwaway cycle. The amount of each nuclide used in the migration calculations is summarized in Table C-9, along with half-life and retardation coefficient. Two additional types of fuel processing options are also listed: uranium oxide recycle and mixed-oxide recycle (Cases 2 and 3 of Task A report). The nuclides tested include four actinide chains and the fission products that are important from the standpoint of amounts of each initially present, as well as of potential health effects.

The parameters used for Scenarios 1 and 2 are summarized in Table C-10. A comment should be made regarding "normal" adsorption. The sorption levels used were taken from the estimated distribution coefficients (K_d) in typical desert soil shown in Table C-3. These values have been compared with measurements for clay, sandstone, caprock, and river sand soil types (Table C-4) and, despite some differences, there are no consistent trends for either actinides or fission products. In some cases, the distribution coefficients in sandstone are higher than the equivalent retardation factors listed in Table C-9, and in other cases they are lower. The sorption distribution coefficient (K_d), rather than the retardation factor (R_d), has been reduced to 10% of the normal values for the sensitivity analysis. (See equations 7 and 8 in Section C-3.1.2 for the relationship between distribution coefficient and retardation factor.) In the case of Tc-99, the evidence supports a relatively high adsorption value ($K_d \geq$ several hundred)⁴² as well as the zero value given in Table C-3. Corresponding R_d values of 3300 and 1.0 have been used for illustrative purposes in calculating concentrations of this nuclide.

TABLE C-10

PARAMETER VALUES USED IN SCENARIOS 1 AND 2

<u>Variable</u>	<u>Aquifer</u>	<u>Communication Area</u>
Thickness, m (ft)	15 (50)	150 (500)
Permeability, cm/sec (ft/day)	10^{-4} (2.8×10^{-1})	10^{-6} (2.8×10^{-3})
Flow gradient, m/m (ft/ft)	0.1	0.1
Adsorption (K_d in ml/g)	Normal	0
Dispersivity, m (ft)	30 (100)	30 (100)
Porosity %	15	1

The grid used in the mathematical model consisted of five 30-m (100-ft) blocks representing the repository and interbeds and 150 52-m (176-ft) blocks representing the aquifer. Potential discharge points were examined in the aquifer at a point above the repository and at each mile distant from the repository. Both the concentration of each nuclide and the aquifer flow rate were summarized at each of these potential discharge points.

C-4.4.3.2 Scenario 2 Results

Migration of actinides and fission products as a function of time and space for the worst probable case (Scenario 2) are summarized in Tables C-11 and C-12. The four actinide chains are shown in Table C-11 for 300, 500, 6000, 30,000, and 100,000 years. Selected fission products are shown in Table C-12 for 300, 500, and 1000 years.

Two leach rate models have been used. One assumes a "constant leach rate", with a correction for the changing surface-to-mass ratio as a function of time. The resulting leach rate decreases approximately exponentially with time. In deriving this leach model, a very conservative leach rate was assumed, in order to account for early temperature effects and possible glass devitrification later.

The second, "variable leach rate", model was a modification of the first, used to define a more severe case during early time. The possible temperature effects were accounted for by increasing the leach rate for the first 4000 years, thus spanning the most important period with respect to fission products. After 4000 years, a lower leach rate was used to account for the probability that devitrification might not occur over the leach time. The comparison of the fraction leached as a function of time was given earlier in Figure C-12. Although the waste form might break down more rapidly than the leach rate model indicates, solubility would then undoubtedly limit a number of the actinide concentrations in the water leaving the repository. This effect was investigated in another model analysis.

TABLE C-11

SCENARIO 2--WORST-CASE FLOW
ACTINIDES AND DAUGHTERS AFTER 300 YEARS

<u>Nuclide</u>	<u>Concentration in g/l at</u> <u>0 km (mile)*</u>
Cm-246	1.1×10^{-10}
Am-242m	1.4×10^{-13}
Pu-242	5.9×10^{-7}
U-238	8.8×10^{-4}
Pu-238	1.8×10^{-8}
U-234	2.8×10^{-7}
Th-230	6.4×10^{-11}
Ra-226	8.6×10^{-12}
Cm-245	9.7×10^{-10}
Pu-241	2.3×10^{-11}
Am-241	1.1×10^{-6}
Np-237	1.0×10^{-4}
U-233	7.5×10^{-11}
Th-229	1.6×10^{-14}
Cm-247	1.6×10^{-12}
Am-243	1.2×10^{-7}
Cm-243	7.9×10^{-14}
Pu-239	7.1×10^{-6}
U-235	7.5×10^{-6}
Cm-248	1.1×10^{-13}
Pu-244	1.8×10^{-17}
Pu-240	2.9×10^{-6}
U-236	3.9×10^{-6}
Th-232	1.1×10^{-11}

*Concentrations at other discharge points are negligible (less than one atom per liter).

TABLE C-11
(Continued)

SCENARIO 2--WORST-CASE FLOW
ACTINIDES AND DAUGHTERS AFTER 500 YEARS

Nuclide	0 km (mile)	Concentration, (g/liter) at Following Distances from Aquifer Entry*				
		1.6 km (1 mile)	3.2 km (2 miles)	4.8 km (3 miles)	6.4 km (4 miles)	8.0 km (5 miles)
Cm-246	3.9×10^{-10}					
Am-242m	1.8×10^{-13}					
Pu-242	2.2×10^{-6}					
U-238	3.2×10^{-3}					
Pu-238	2.4×10^{-8}					
U-234	1.0×10^{-6}					
Th-230	4.1×10^{-10}					
Ra-226	8.4×10^{-11}					
Cm-245	3.4×10^{-9}					
Pu-241	6.3×10^{-12}					
Am-241	3.0×10^{-6}					
Np-237	3.0×10^{-4}	2.2×10^{-14}		7.0×10^{-25}		
U-233	4.2×10^{-10}	1.0×10^{-20}				
Th-225	1.6×10^{-13}	2.4×10^{-24}				
Cm-247	5.7×10^{-12}					
Am-243	4.1×10^{-7}					
Cm-243	1.1×10^{-15}					
Pu-239	2.6×10^{-5}					
U-235	2.8×10^{-5}					
Cm-248	4.0×10^{-13}					
Pu-244	1.1×10^{-16}					
Pu-240	1.0×10^{-5}					
U-236	1.4×10^{-5}					
Th-232	6.3×10^{-11}					

*Concentrations at other discharge points are negligible
(less than one atom per liter).

TABLE C-11
(Continued)

SCENARIO 2--WORST-CASE FLOW
ACTINIDES AND DAUGHTERS AFTER 6000 YEARS

Nuclide	0 km (mile)	Concentration, (g/liter) at Following Distances from Aquifer Entry*				
		1.6 km (1 mile)	3.2 km (2 miles)	4.8 km (3 miles)	6.4 km (4 miles)	8.0 km (5 miles)
Cm-246	5.0×10^{-10}					
Am-242m						
Pu-242	8.1×10^{-6}					
U-238	1.3×10^{-2}					
Pu-238	3.9×10^{-12}					
U-234	4.1×10^{-6}					
Th-230	2.1×10^{-8}					
Ra-226	1.3×10^{-8}	4.4×10^{-18}				
Cm-245	2.8×10^{-9}	2.8×10^{-24}				
Pu-241	1.4×10^{-12}					
Am-241	6.3×10^{-9}					
Np-237	2.4×10^{-5}	7.4×10^{-5}	5.0×10^{-5}	1.5×10^{-5}	3.1×10^{-6}	4.9×10^{-7}
U-233	1.9×10^{-8}	1.8×10^{-9}	6.7×10^{-10}	1.6×10^{-10}	2.8×10^{-11}	4.1×10^{-12}
Th-229	1.6×10^{-10}	9.7×10^{-12}	2.8×10^{-12}	5.8×10^{-13}	9.6×10^{-14}	1.4×10^{-14}
Cm-247	1.1×10^{-11}					
Am-243	5.3×10^{-7}					
Cm-243						
Pu-239	2.0×10^{-1}					
U-235	6.4×10^{-2}					
Cm-248	7.3×10^{-13}					
Pu-244	6.6×10^{-15}					
Pu-240	1.2×10^{-5}					
U-236	7.8×10^{-5}					
Th-232	8.9×10^{-9}					

*Blank indicates extremely small concentration, less than one atom per liter.

TABLE C-11
(Continued)

SCENARIO 2--WORST-CASE FLOW
ACTINIDES AND DAUGHTERS AFTER 30,000 YEARS

Nuclide	0 km (mile)	Concentration, (g/liter) at Following Distances from Aquifer Entry*				
		1.6 km (1 mile)	3.2 km (2 miles)	4.8 km (3 miles)	6.4 km (4 miles)	8.0 km (5 miles)
Cm-246	4.9×10^{-12}					
Am-242m						
Pu-242	5.5×10^{-6}					
U-238	1.1×10^{-2}					
Pu-238						
U-234	3.4×10^{-6}					
Th-230	9.3×10^{-8}					
Ra-226	6.6×10^{-8}	8.0×10^{-14}				
Cm-245	5.2×10^{-10}	1.2×10^{-21}				
Pu-241	2.7×10^{-13}					
Am-241	1.2×10^{-11}	1.9×10^{-23}				
Np-237	1.1×10^{-5}	1.8×10^{-5}	4.2×10^{-5}	5.5×10^{-5}	3.7×10^{-5}	1.6×10^{-5}
U-233	3.6×10^{-8}	2.7×10^{-9}	2.3×10^{-9}	1.5×10^{-9}	6.6×10^{-10}	2.3×10^{-10}
Th-229	4.2×10^{-10}	2.5×10^{-11}	1.7×10^{-11}	8.7×10^{-12}	3.4×10^{-12}	1.0×10^{-12}
Cm-247	4.9×10^{-12}	8.1×10^{-21}				
Am-243	1.4×10^{-7}					
Cm-243						
Pu-239	8.4×10^{-2}	2.9×10^{-23}				
U-235	7.1×10^{-2}					
Cm-248	3.4×10^{-13}					
Pu-244	8.7×10^{-15}					
Pu-240	2.8×10^{-6}					
U-236	7.3×10^{-5}					
Th-232	1.8×10^{-8}					

*Blank indicates extremely small concentration,
less than one atom per liter.

TABLE C-11
(Continued)

SCENARIO 2--WORST-CASE FLOW
ACTINIDES AND DAUGHTERS AFTER 100,000 YEARS

Nuclide	0 km (mile)	Concentration, (g/liter) at Following Distances from Aquifer Entry*				
		1.6 km (1 mile)	3.2 km (2 miles)	4.8 km (3 miles)	6.4 km (4 miles)	8.0 km (5 miles)
Cm-246	4.3×10^{-18}	2.8×10^{-21}				
Am-242m						
Pu-242	6.7×10^{-7}	1.2×10^{-19}				
U-238	2.7×10^{-3}	3.3×10^{-20}				
Pu-238						
U-234	7.1×10^{-7}					
Th-230	9.5×10^{-8}					
Ra-226	6.9×10^{-8}	1.6×10^{-12}	2.1×10^{-18}			
Cm-245	6.4×10^{-14}	7.8×10^{-17}				
Pu-241	3.3×10^{-17}	4.0×10^{-20}				
Am-241	1.3×10^{-15}	1.4×10^{-18}				
Np-237	5.7×10^{-14}	1.1×10^{-9}	1.1×10^{-7}	8.7×10^{-7}	2.5×10^{-6}	4.2×10^{-6}
U-233	2.2×10^{-8}	2.7×10^{-9}	2.7×10^{-9}	2.8×10^{-9}	2.9×10^{-9}	2.9×10^{-9}
Th-229	3.6×10^{-10}	3.5×10^{-11}	3.6×10^{-11}	3.6×10^{-11}	3.7×10^{-11}	3.6×10^{-11}
Cm-247	5.9×10^{-14}	2.8×10^{-15}				
Am-243	9.4×10^{-11}	2.4×10^{-19}				
Cm-243						
Pu-239	1.1×10^{-3}	1.4×10^{-15}				
U-235	1.9×10^{-2}	1.4×10^{-15}				
Cm-248	3.7×10^{-15}	1.6×10^{-16}				
Pu-244	2.7×10^{-15}	1.1×10^{-18}				
Pu-240	1.1×10^{-9}	5.6×10^{-23}				
U-236	1.8×10^{-5}	9.1×10^{-21}				
Th-232	3.3×10^{-8}					

* Blank indicates extremely small concentration, less than one atom per liter.

Note: This table includes four separate actinide series. Nuclides omitted from the above results should be assumed to be in secular equilibrium with their parent component, i.e., the activity of each nuclide would be equal. For example, an equilibrium concentration of Rn-222 would be present along with the values listed for Ra-226.

TABLE C-12

SCENARIO 2--FISSION PRODUCTS AND C-14

Time, yr	Nuclide	0 km (mile)	Concentration, (g/liter) at Following Distances from Aquifer Entry*				
			1.6 km (1 mile)	3.2 km (2 miles)	4.8 km (3 miles)	6.4 km (4 miles)	8.0 km (5 miles)
300	C-14	1.7×10^{-9}					
	Sr-90	2.6×10^{-8}					
	Tc-99R [†]	6.7×10^{-11}					
	Tc-99N	5.1×10^{-4}	1.3×10^{-4}	5.8×10^{-7}	8.7×10^{-11}	1.8×10^{-15}	9.8×10^{-21}
	I-129	1.4×10^{-4}	3.7×10^{-5}	1.6×10^{-7}	2.4×10^{-11}	4.9×10^{-16}	2.7×10^{-21}
	Cs-137	1.2×10^{-8}					
500	Sm-151	2.0×10^{-8}					
	C-14	1.3×10^{-9}	1.2×10^{-14}				
	Sr-90	5.3×10^{-10}					
	Tc-99R	2.4×10^{-10}					
	Tc-99N	3.2×10^{-4}	3.6×10^{-4}	4.7×10^{-4}	3.6×10^{-4}	6.9×10^{-5}	2.0×10^{-6}
	I-129	8.7×10^{-5}	1.0×10^{-4}	1.3×10^{-4}	1.0×10^{-4}	1.9×10^{-5}	5.5×10^{-7}
1000	Cs-137	4.8×10^{-10}					
	Sm-151	1.6×10^{-8}					
	C-14	5.6×10^{-10}	7.1×10^{-10}	1.9×10^{-14}	5.7×10^{-23}		
	Sr-90	3.2×10^{-15}					
	Tc-99R	4.6×10^{-10}					
	Tc-99N	1.5×10^{-4}	1.6×10^{-4}	1.8×10^{-4}	2.0×10^{-4}	2.3×10^{-4}	2.6×10^{-4}
	I-129	4.2×10^{-5}	4.5×10^{-5}	5.0×10^{-5}	5.6×10^{-5}	6.3×10^{-5}	7.2×10^{-5}
	Cs-137	1.4×10^{-14}					
	Sm-151	6.6×10^{-10}					

*Blank indicates extremely small concentrations, less than one atom per liter.

[†]Tc-99R represents an assumed chemical form of Tc-99 that is highly sorbed ($R_d=3300$).

Tc-99N represents a non-sorbed chemical form ($R_d=1$).

This second leach model is represented by the following equation:

$$\text{Fraction leached} = 0.173 \ln (1 + t/200) \quad (14)$$

where

t = time in years after first contact with water (200 years).

This equation shows that all the glass is leached at 65,000 years, with the leaching period continuing over most of the time interval shown.

The rate, however, decreased substantially from the start of leaching at 200 years until the end of leaching at 65,000 years.

The results summarized in Table C-11 show significant amounts of several nuclides entering the aquifer at 6000 years. For example, U-238 and Pu-242 would have 1900 g/day and 1.2 g/day, respectively. These rates were obtained by multiplying the concentrations, 1.3×10^{-2} g/liter and 8.1×10^{-6} g/liter, respectively, by the water flow rate in the aquifer, 1.44×10^5 liter/day. The concentration of U-238 of 1.3×10^{-2} g/liter translates to 4300 picocuries/liter (pCi/liter) and of Pu-242 to 47,000 pCi/liter. However, significant levels of these nuclides never reach a point 1.6 km (1 mile) from the repository even after 100,000 years or more. The worst actinide migration occurs with Np-237. About 3.5 g/day are still reaching the aquifer at 6000 years. The peak potential discharge rate of Np-237 at a distance 1.6 km (1 mile) from the repository would be about 10 g/day; at 3.2 km (2 miles), about 9 g/day; and at 4.8 km (3 miles), about 8 g/day. Because U-233 and Th-229 are daughters of Np-237, concentrations of these nuclides are found to have moved substantially farther than would have been expected.

The migration of fission products is summarized in Table C-12 for times of 300, 500, and 1000 years. Significant amounts of I-129 are present and are not adsorbed; hence this nuclide has substantial migration. Table C-13 lists the maximum value of I-129 in both g/liter and pCi/liter, calculated at various distances from the repository.

In Scenario 2 permeability of the repository strata was increased by two orders of magnitude. The highest interstitial horizontal permeability listed for this strata was 10^{-7} cm/sec. The corresponding vertical

TABLE C-13

PEAK VALUES OF I-129

<u>Distance from Repository -km (miles)</u>	<u>1.6 (1)</u>	<u>3.2 (2)</u>	<u>4.8 (3)</u>	<u>6.4 (4)</u>	<u>8.0 (5)</u>
Concentration, g/liter	1.34×10^{-4}	1.30×10^{-4}	1.27×10^{-4}	1.25×10^{-4}	1.20×10^{-4}
pCi/liter	21,840	21,190	20,700	20,380	19,560
Time (yr)	389	470	551	632	700

permeability was 10^{-6} cm/sec. This case could be thought of as representing a substantial fracture zone above the repository, with a uniform set of fractures spaced 3m (10 ft) apart and with a fracture width of 1.5 cm (0.05 ft); the relative cross-sectional area is then approximately $(0.025)(10)(4)/100 = 0.01$. If these fractures had a permeability of 10^{-4} cm/sec, the communication area above the repository would yield a net permeability of about 10^{-6} cm/sec and a net porosity of about 1%, which were the values used in Scenario 2. This scenario simply sets out a conceptual illustration of what has been simulated--namely, a "worst case" geosphere migration, since a fracture area has created the communication pathway.

C-4.4.3.3 Scenario 1 Results

Permeability and flow gradient are important variables controlling the base flow in the aquifer. Probably more important to the geosphere migration pathway, however, is the permeability of the communication area between the repository and the aquifer. Section C-4.1 discussed the magnitude of the natural flow that might occur in potential repository strata. These velocities would be quite low and probably not important from the standpoint of nuclide migration. More important would be those events that could lead to a more permeable area occurring between the aquifer and the repository:

- (1) Breakdown of the shaft entry seal over long time periods;
- (2) Thermally-induced fractures around the repository caused by heat from radioactive decay;
- (3) Seismic activity.

In Scenario 2 a rather permeable area above the repository was assumed. The porosity for the repository strata was assumed to be 1% and thus vertical migration rates included in Scenario 2 calculations are indicative of fractured media. This flow would undoubtedly exceed the actual flow, even considering the thermal convection effects.

Scenario 1 is nearly identical to Scenario 2, except that in order to examine a low end of the range for vertical migration rates of nuclides from the repository, the strata fluids were assumed to have zero vertical velocity. Liquid contact was postulated, but nuclides were assumed to be

transported from the repository only by molecular diffusion in the liquid phase. A net diffusion coefficient equal to 10^{-6} cm²/sec was used in this calculation. This net diffusivity is made up of an open-stream molecular diffusion coefficient of 5×10^{-6} cm²/sec and a tortuosity factor equal to five.

The diffusion pathway in the repository strata should represent a lower bound on the transport of nuclides from the repository to the aquifer (unless these strata are also strongly adsorbing). However, in the absence of a solubility limitation for the nuclides, even a diffusional concentration gradient will increase until the amount of nuclides diffusing from the repository equals the leach rate. Although there is a much longer time transient involved before the nuclide fluxes into the aquifer are equal to the leach rate, the amounts entering the aquifer by either diffusion or flow would eventually be the same--in the absence of a solubility limitation. Calculations have been made to ascertain how long this time transient might be.

The results are tabulated in Table C-14 for the actinide series at 6000 and 30,000 years. Np-237 has a concentration at the 0 km (mile) location of about two orders of magnitude less than the Scenario 2 flow at 6000 years and about a factor of five less at 30,000 years. Np-237 was selected for the comparison because of its low adsorptivity and its relatively large contribution to dose.

The fission product results for the diffusion pathway (Scenario 1) are tabulated in Table C-15. In this case, I-129 has a concentration at the 0 location of about eight orders of magnitude less than in Scenario 2 at 500 years, and six orders less at 1000 years. In Scenario 2, the peak I-129 concentration had already gone beyond the 6.4 km (4-mile) migration point in the aquifer at 1000 years. At 1000 years in Scenario 1, the concentrations are still increasing, but are nearly at their peak values. The condition results from an interaction between the increasing diffusion gradient and the decreasing leach rate.

These results indicate that over the time periods of interest for either fission products or actinides, the concentrations migrating in an aquifer would be at least a factor of five lower for the diffusion pathway

TABLE C-14

SCENARIO 1--DIFFUSION PATHWAY - ACTINIDES AND DAUGHTERS AFTER 6000 YEARS

Nuclide	0 km (mile)	Concentrations (g/liter) at Following Distances from Aquifer Entry*				
		1.6 km (1 mile)	3.2 km (2 miles)	4.8 km (3 miles)	6.4 km (4 miles)	8.0 km (5 miles)
Cm-246	4.0×10^{-13}					
Am-242m						
Pu-242	6.8×10^{-9}					
U-238	1.0×10^{-5}					
Pu-238	1.3×10^{-20}					
U-234	3.4×10^{-9}					
Th-230	2.0×10^{-11}					
Ra-226	1.7×10^{-11}	6.2×10^{-24}				
Cm-245	5.9×10^{-12}					
Pu-241	2.8×10^{-15}					
Am-241	3.4×10^{-12}					
Np-237	3.2×10^{-7}	4.4×10^{-9}	5.0×10^{-12}			
U-233	3.5×10^{-11}	1.1×10^{-14}	9.9×10^{-16}			
Th-229	1.5×10^{-13}	1.3×10^{-17}	9.5×10^{-21}			
Cm-247	1.6×10^{-14}					
Am-243	6.8×10^{-10}					
Cm-243						
Pu-239	6.7×10^{-8}					
U-235	1.0×10^{-7}					
Cm-248	1.2×10^{-15}					
Pu-244	4.7×10^{-18}					
Pu-240	1.6×10^{-8}					
U-236	5.8×10^{-8}					
Th-232	3.4×10^{-12}					

* Blank indicates extremely small concentration, less than one atom per liter.

TABLE C-14
(Continued)

SCENARIO 1--DIFFUSION PATHWAY - ACTINIDES AND DAUGHTERS AFTER 30,000 YEARS

Nuclide	0 km (mile)	Concentrations (g/liter) at Following Distances from Aquifer Entry*				
		1.6 km (1 mile)	3.2 km (2 miles)	4.8 km (3 miles)	6.4 km (4 miles)	8.0 km (5 miles)
Cm-246	3.5×10^{-13}					
Am-242m						
Pu-242	2.7×10^{-7}					
U-238	4.7×10^{-4}					
Pu-238						
U-234	1.4×10^{-7}					
Th-230	3.7×10^{-9}					
Ra-226	2.8×10^{-9}	5.5×10^{-16}	2.7×10^{-24}			
Cm-245	2.8×10^{-11}					
Pu-241	1.4×10^{-14}					
Am-241	5.3×10^{-13}					
Np-237	2.3×10^{-6}	2.0×10^{-6}	1.3×10^{-6}			
U-233	5.5×10^{-9}	5.0×10^{-11}	2.2×10^{-11}			
Th-229	6.1×10^{-11}	2.9×10^{-13}	1.0×10^{-13}			
Cm-247	4.4×10^{-13}					
Am-243	3.4×10^{-9}					
Cm-243						
Pu-239	1.5×10^{-6}					
U-235	5.5×10^{-6}					
Cm-248	3.0×10^{-14}					
Pu-244	7.2×10^{-16}					
Pu-240	6.3×10^{-8}					
U-236	3.1×10^{-6}					
Th-232	8.2×10^{-10}					

* Blank indicates extremely small concentration, less than one atom per liter.

TABLE C-15

SCENARIO 1--DIFFUSION PATHWAY - FISSION PRODUCTS AND C-14

		Concentration (g/liter) at Following Distances from Aquifer Entry*				
Nuclide	0 km (mile)	1.6 km (1 mile)	3.2 km (2 miles)	4.8 km (3 miles)	6.4 km (4 miles)	8.0 km (5 miles)
<u>300 Years</u>						
C-14	4.3×10^{-19}	1.3×10^{-25}				
Sr-90	1.2×10^{-19}					
Tc-99R†	2.4×10^{-20}					
Tc-99N	1.6×10^{-13}	3.1×10^{-13}	4.7×10^{-14}			
I-129	4.4×10^{-14}	8.6×10^{-14}	1.3×10^{-14}			
Cs-137	7.2×10^{-20}					
Sm-151	1.4×10^{-18}					
<u>500 Years</u>						
C-14	1.4×10^{-17}	4.4×10^{-22}				
Sr-90	4.4×10^{-20}					
Tc-99R	2.7×10^{-18}					
Tc-99N	4.7×10^{-12}	1.9×10^{-12}	6.9×10^{-13}			
I-129	1.3×10^{-12}	5.3×10^{-13}	1.9×10^{-13}			
Cs-137	4.0×10^{-20}					
Sm-151	1.5×10^{-17}					
<u>1000 Years</u>						
C-14	7.6×10^{-16}	1.4×10^{-17}	6.8×10^{-20}			
Sr-90	3.3×10^{-24}					
Tc-99R	6.8×10^{-17}					
Tc-99N	2.4×10^{-10}	1.7×10^{-10}	1.2×10^{-10}			
I-129	6.5×10^{-11}	4.7×10^{-11}	3.3×10^{-11}			
Cs-137	8.7×10^{-24}					
Sm-151	1.4×10^{-17}					

* Blank indicates extremely small concentration, less than one atom per liter.

† Tc-99R represents an assumed chemical form of Tc-99 that is highly sorbed ($R_d = 3300$).

Tc-99N represents a non-sorbed form of Tc-99 ($R_d = 1$).

(Scenario 1) than for water flow through the repository (Scenario 2). Because of the high concentration levels necessary for the diffusional transport to be effective, solubility limitations will also be important and will further reduce the concentration reaching an aquifer.

C.4.4.3.4 Sensitivity Analyses Using the Numerical Model

C-4.4.3.4.1 Effect of Leach Rate, Containment, and Nuclide Solubility

For the actinides, solubility limits the actual leach rate below that utilized in Scenario 2. In order to examine this effect, the leach model used in Scenario 2 includes the limitation that if a nuclide concentration exceeds its solubility, only the soluble portion leaves the repository. This was accomplished by including a material balance of nuclides in the repository. These nuclides could exist in the solid waste form (assumed here to be glass), in soluble form, or in an insoluble form (precipitate) after the glass matrix has dissolved. The case chosen for illustration is the actinide series Cm-246 and Am-242m. The solubility values used to limit these components are given in Table C-16.⁽⁴³⁾ The figures taken were for the most soluble form of each nuclide at a neutral pH in fresh water, since this condition gave the highest solubility values.

The results are quite dramatic. The glass was completely leached at 65,000 years, but even at 100,000 years substantial amounts of uranium and plutonium remained in an insoluble form in the repository. These nuclides would continue to leach very slowly over long periods of time. In Scenario 2, 1900 g/day of U-238 reached the aquifer at a time of 6000 days; when the solubility limit was imposed, however, less than 3×10^{-8} g/day reached the aquifer. There is a similar, but not as drastic, reduction in the concentration of plutonium reaching the aquifer. The solubility limit will undoubtedly be an important influence in controlling the leaching rate of the actinides.

The migration of fission products should be most sensitive to the leach rate and to the effectiveness of containment. In order to investigate

TABLE C-16
ACTINIDE SOLUBILITY LIMITS⁽⁴³⁾

<u>Actinide</u>	<u>Sat'd NaCl</u> <u>(g-moles/liter)</u>	<u>Water</u> <u>(g-moles/liter)</u>
U	$\log [\text{UO}_2 \text{Cl}^+] = -28.7$	$\log [\text{UO}_2^{++}] = -9$
N _p	$\log [\text{N}_p \text{O}_2^+] = -3$	$\log [\text{N}_p \text{O}_2^+] = -3$
Pu	$\log [\text{PuO}_2 \text{Cl}] = -11.8$	$\log [\text{Pu O}_2^+] = -12.3$
Am	$\log [\text{Am OH}^{++}] = -18.4$	$\log [\text{Am OH}^{++}] = -18.4$

Cm values were taken to be similar to those for Pu.

this effect, a case was assumed where the important fission products were completely leached in a 25-year period. Leaching was started 50 years after the repository was closed. The results can be contrasted to Scenario 2 in which the maximum I-129 concentration was about 1.3×10^{-4} g/liter at a distance 1 mile from the repository (see Table C-13). With the more rapid leaching rate and shorter containment period, the I-129 concentration at this same distance peaked at a concentration more than 20 times higher, 2.5×10^{-3} g/liter. The concentration of Sr-90 entering the aquifer never exceeded 3×10^{-8} g/liter in Scenario 2, but with the shorter leach time and containment, Sr-90 reached a level nearly 10^4 times higher at 2×10^{-4} g/liter. Sr-90 still would not present a problem, however, once migration into the aquifer takes place because of its high adsorption coefficient.

In Scenario 2, a standard glass form with leaching starting 200 years after emplacement was assumed; water contact was present when the repository was sealed and the containment then broke down in 200 years. The "variable leach rate" model used gave higher leach rates than the "constant leach rate" model for periods up to 4000 years but then leaching declined. The total leach time was 65,000 years. (See Figure C-12).

A set of sensitivity calculations was performed using the same conditions as in Scenario 2, except that the "constant leach rate" model was used. In this instance, the waste form leached completely in 20,000 years. Nevertheless, the results showed only small variances in nuclide concentrations from those resulting from the assumptions of Scenario 2. At a similar discharge point no nuclide concentrations differed by more than a factor of 2. The fission product concentrations were lower (as was expected) since the crossover point in leach rates is 4000 years; the concentrations of radium-226 and neptunium-237 were higher for the "constant leach rate" model than for the "variable leach rate" model, but by less than a factor of 2.

As mentioned previously, solubility would undoubtedly limit the actual leach rate of actinides in either case. Thus, the ultimate difference between the two leach models would be even smaller.

Another factor associated with the source terms is the method of reprocessing, if any, of the waste form before it is placed in the repository. Three alternatives were discussed in Task A: (1) no reprocessing, i.e., the "throwaway" cycle; (2) uranium recycle only (plutonium is also separated but not recycled), and (3) mixed uranium and plutonium oxide recycle (see Table A-3). In the latter two cases, the important radionuclides removed are plutonium, uranium, and iodine. Slight reductions in other nuclides also occur in these cases where recycling is used. A few of the heavy nuclide concentrations are increased, Am-243 perhaps being most important. The relative hazard from a recycling case should always be considerably less than in a "throwaway" case, and thus no geosphere calculations were made for the recycle cases. An important reduction in the amount of I-129 from that initially present is obvious from Table C-8. This reduction of nearly three orders of magnitude in the initial amount present in the waste form should decrease the I-129 concentrations along the geosphere pathway proportionately. The nuclides whose quantities have increased from the amounts initially present in the waste form are all actinides; from the previous calculations it appears that solubility would limit concentrations of these nuclides in the geosphere transport pathway.

C-4.4.3.4.2 Effect of Adsorption

One highly uncertain factor is the adsorptive character of the formation, including both the repository strata and interbeds, and the aquifer. In Scenario 2, normal adsorption in the aquifer and none in the communication area was assumed. In order to examine the importance of adsorption, computer runs were made using different sorption values.

In a low-sorption case, the aquifer was assumed to have much less adsorption than in either Scenarios 1 or 2. The actinide chain beginning with curium-246 was used in this illustration. Adsorption values based on one-tenth the original distribution coefficients (Table C-9) were used for the aquifer. Again, no adsorption was assumed in the repository or in the communicating area.

The results for this case indicate that even with much lower adsorption in the aquifer, the actinide series migration will be effectively retarded. The two components of most concern from this actinide chain would be radium-226 and plutonium-242. At a distance of 1.6 km (1 mile) from the repository at 6000 years, the concentration of Ra-226 had increased to 1.9×10^{-9} g/liter (2000 pCi/l), which is a large increase (almost nine orders of magnitude) over that of Scenario 2. Pu-242 concentration had also undergone a large relative increase, but was still extremely small. It should be emphasized that under idealized conditions of a constant leach rate and virtually no decay, the adsorption level in the aquifer primarily affects time of arrival rather than the concentration level of a nuclide. In the above cases, the effect is larger, since the leach rate is decreasing with time and the nuclides are either decaying or being formed. As a result, the concentrations can be affected substantially.

In the high-sorption case, the communicating area was assumed to have substantial adsorption (as it would if the repository were placed in shale). In this case, twice the original distribution values were used in the communication area. Normal adsorption was used in the aquifer.

In this case, substantial decreases in nuclide concentrations resulted from strong adsorption in the repository strata. The residence time of components, such as Ra-226, increased to 50,000 years compared with roughly 50 years in the no-adsorption condition of Scenario 2. Decay, dispersion, and leach-rate changes are important over this increased residence time.

C-4.5 COMPARISON OF THE EFFECT OF LOW-GRADE URANIUM ORE VS. AN HLW REPOSITORY

For this comparison of the HLW with a natural uranium ore body, the assumption is made that the site of the HLW is occupied by a 0.2% uranium ore body, which is in the same location in the ground and is subject to the same influences and conditions as assumed for Scenario 2.

For the calculation, the natural ore is assumed to consist of U-238 and its important daughter components contained in the repository volume (300 acres by 15 m [50 ft] thick). After several initial calculations, four nuclides (U-238, U-234, Th-230, and Ra-226) were found to be sufficient to represent the entire decay chain. The amount of each nuclide leaving the ore was limited by its solubility. The limiting solubilities are as follows:

<u>Nuclide</u>	<u>Limiting Solubility, moles/l</u>
U-238	1×10^{-9} (oxide) ⁽⁴³⁾
U-234	1×10^{-9} (oxide) ⁽⁴³⁾
Th-230	1×10^{-9} (hydroxide) ⁽⁴⁴⁾
Ra-226	7×10^{-5} (carbonate)*

For comparative purposes, the aquifer and flow through the ore deposit were assumed to be the same as for the Scenario 2 repository calculation.

There is a distinct difference between nuclide transport from the ore and that from an HLW repository. In the HLW, there is an actinide decay chain, beginning with Cm-246, which contains U-238 as a daughter. This decay chain also produces Ra-226 by a separate route that does not involve U-238. As a consequence, in an intact repository from which no nuclides have escaped, Ra-226 attains its maximum concentration between 100,000 and 200,000 years after the HLW has been removed from a reactor. The amount of Ra-226 would then decrease slowly toward the equilibrium value produced by the decay of U-238 alone.

*Estimated by analogy to the solubility of barium carbonate.⁽⁴⁴⁾

In the ore deposit, the amount of Ra-226 initially present would be only that quantity produced by the decay of the U-238. However, since the ore would not be confined by the kinds of barriers used in an HLW repository, Ra-226 would be removed rapidly because of solubility in water and the resulting equilibrium concentration of Ra-226 would be controlled only by its rate of formation from decay of U-238.

For a moderately low-grade ore deposit (0.2% U-238), the area equivalent to a repository would contain about the same mass of U-238 as does the throwaway cycle HLW. In this analysis, these amounts are 4.72×10^{10} g for HLW and 7×10^{10} g for the ore deposit. The following table summarizes the Ra-226 concentrations arriving at various distances in the aquifer.

Distance:	Time, (years)	Ra-226 Concentrations (g/liter)	
		From Ore	From HLW
0 km (0 mile)	6,000	7.1×10^{-8}	1.3×10^{-8}
	30,000	7.3×10^{-8}	6.6×10^{-8}
	100,000	7.3×10^{-8}	6.6×10^{-8}
1.6 km (1 mile)	6,000	2.5×10^{-17}	4.4×10^{-18}
	30,000	2.1×10^{-13}	8.0×10^{-14}
	100,000	2.6×10^{-13}	1.8×10^{-12}

In the above comparison, the concentrations of Ra-226 from the ore deposit are quite similar to those from the HLW; directly above the repository, the concentrations are virtually the same. At 1.6 km (1 mile) distance, however, the concentrations of Ra-226 from the HLW are smaller than those from the ore for times up to 30,000 years but become larger after that. The reason for this is that the HLW contains substantial initial inventories of Pu-238 and U-234 (both parent nuclides of Ra-226). As these parent nuclides decay, significant quantities of Ra-226 are created. In the ore deposit, only U-238 decay produces Ra-226; thus, the concentrations at later times approach a constant value. The magnitude of this equilibrium concentration depends primarily on the travel time and upon the decay rate of Ra-226.

One additional factor should be mentioned. In the above analysis, water was assumed to have come in contact with the ore (or with the HLW) at the moment calculation was started. It is equally reasonable to assume that water would have been flowing through the ore continuously over geologic time. In that event, the aquifer might be saturated with respect to adsorption of U-238 and its daughter products. This "saturation" would then cause water from this aquifer to have essentially a constant ratio of U-238/Ra-226. This is in reasonable agreement with an extensive data set collected by the U.S. Geological Survey.⁽⁴⁵⁾ The largest number of samples typically had concentrations of 1×10^{-6} g/l U-238 and 1×10^{-12} g/l of Ra-226. For distances beyond roughly 1.6 km (1 mile) from the ore, the U-238 that is first dissolved in the water and then adsorbed downstream can produce more Ra-226 in situ than is formed in the ore and then transported to the same point.

C-5:0 DOSE-TO-MAN CONSIDERATIONS

C-5.1 OBJECTIVES AND OVERVIEW

C-5.1.1 Introduction and Purpose

A repository for high-level radioactive wastes will contain large quantities of radionuclides in high concentrations. Placement of the wastes in deep geologic formations and in physical forms and containment structures designed to minimize the chances of escape and transport in the environment is being considered as a way of protecting the public from the potential hazards associated with such wastes.

The objective of this section of the Task C report is to provide a basis on which to estimate the magnitudes of radiation doses that might result from releases at any time in the future. More specifically, this part of the project provides estimates of doses to individuals and populations as a result of radionuclides entering various environmental media and being transported through the biosphere via likely pathways. A further objective of this task is to compare the calculated radiation doses resulting from the potential repository leakage with radiation dose guides applicable to the public and with radiation doses resulting from naturally-occurring radionuclides, and to assess the importance of these predicted radiation doses.

Because of the long half-lives of some of the radionuclides that would be placed in a geologic repository, the potential toxicity of those materials persists for long periods. In general, the few nuclides that persist in the repository will be the same as those that occur naturally in the earth, e.g., the uranium, thorium, neptunium, and actinium decay series. The long life of the high-level waste (HLW) necessitates the consideration of impacts on the biosphere, including radiation doses, over equally long time spans. Indeed, the National Environmental Policy Act of 1969 requires that such projections be made.

C-5.1.2 Options and Considerations for Making Radiation Dose Assessments

In assessing the radiation impact of HLW on the public, the procedure is to postulate a pathway by which radioactivity from an HLW repository can get into an individual and to use a metabolic model to calculate the absorbed dose (rad), or preferably the dose equivalent (rem, i.e., the absorbed dose multiplied by a quality factor) to a specific organ or to the whole body of the exposed person(s). The dose equivalent is the important radiological quantity for estimating the health effects resulting from a particular radiation exposure.*

For an individual, therefore, the magnitude, and hence the acceptability or risk, of the radiation impact can be compared to the dose limits recommended by national and international organizations. For the case of a group of persons, i.e., a population, a similar direct comparison is also desirable. This can be done by invoking the concept of a dose equivalent to the population (the unit used is person-rem), consisting of the sum of the dose equivalents to each individual of the population. The collective effect of the dose equivalent to this population is assumed to be independent of the dose equivalent distribution in the population. The condition that permits such a dose equivalent comparison is a linear-nonthreshold dose-response model of the biological effects of radiation. Such a model is mathematically convenient, is unlikely to underestimate the health effects, and permits direct comparisons to dose limits and dose equivalents from other sources, especially when the dose equivalents are at or below the dose limits or are equivalent to those from natural causes. (See Section C-5.1.4 for further discussion of the model and its implications.)

Assessment of the radiation impact can be made by comparing the concentrations in drinking water of radionuclides derived from HLW with the maximum permissible concentrations recommended by national and international organizations. Such a comparison can be useful, especially if drinking water is the main route of entry into persons. However, the recommended concentrations are based on dose limits and again the comparison is really one of dose equivalents.

*While prediction of health effects is a subject that is beyond the scope of this report, it is assumed that dose estimates will eventually be used for this purpose.

Further assessments can be made by comparing the dose equivalents due to the HLW with dose equivalents due to natural background, especially that due to internal radiation from radionuclides such as carbon-14, potassium-40, and the uranium and thorium series.

Comparisons based on relative hazard indices are sometimes made as an alternate approach and to avoid the necessity of making absolute health estimates. Such indices include:

- (1) The total volume of water required to dilute the radioactivity in the HLW to the maximum permissible concentration for drinking water,⁽⁴⁶⁾ or
- (2) The total number of "annual intake limits" contained in the HLW for either the radiation worker or for a member of the general public,⁽⁴⁷⁾ or
- (3) The ultimate number of cancer deaths that might be produced by a given quantity of HLW, if distributed in a large population in such a way as to produce the maximum biological effect.⁽⁴⁸⁾

While these indices may serve as extreme boundary conditions or may be intellectually interesting, they are not very useful and can even be misleading, since they omit all the transport processes and metabolic pathways that affect different radionuclides in different ways in the final calculations of dose equivalent.

It is possible to make a relative comparison between important HLW radionuclides and natural radionuclides, provided that the natural radionuclide is assumed to be placed in the repository along with the other radionuclides. The comparison will not be a proper one if the natural radionuclide is not exposed to the same transport processes as the HLW.

C-5.1.2.1 Prediction of Pathways

The transport processes described in Section C-4 are based on geological considerations that are associated with long time periods in the past and that will be generally applicable for the long time periods into the future associated with HLW disposal. Thus, the transport processes described and discussed at present will presumably be unchanged for equivalent periods in the future.

The analogous situation for predicting the pathways by which the radionuclides will disperse in the biosphere and enter people, and even for predicting human metabolism, does not have the same sound basis of constancy with time. Society and the activities of people have been changing continually and quite rapidly in relation to time periods as long as those involved in HLW management. However, to make a dose equivalent calculation, a pathway and metabolic model must be postulated describing the conditions under which the radionuclides are introduced into people and how they are metabolized. If society and the activities of people are known, realistic models can be postulated and the calculations made for the short term. For the long term assessment, on the other hand, it is nearly impossible to predict what society and the activities of people will be and, hence, no applicable models have been postulated and no appropriate calculations can be made.

This inability to predict pathways and metabolism for the long term poses great limitations for dose calculations. This concept is important enough that it is described in greater detail in Section C-5.1.3 in order to give a broader perspective to the dose equivalent calculations that are made in a more conventional manner and are presented in Section C-5.2 and C-5.3.

C-5.1.2.2 Assumptions and Conditions for the Dose Calculations

In spite of the difficulties of long-term predictions, estimates of dose equivalents are needed to make an evaluation of the potential consequence to populations in the future.

Dose equivalent calculations are expressed in terms of dose commitment. The latter is defined for the present purpose as the dose equivalent accumulated over a period of 50 years following the intake of radionuclides for a period of not more than one year. The dose equivalent, as defined, will vary with the age of the individual and is, therefore, age specific. The representative age chosen for this report is 21 years.

The models for the calculations involve two types of transport from the repository, three types of pathways to people, and three types of dose commitment to these people. The transport scenarios both involve water getting into the repository; in Scenario 1 there is water contact but no flow, and the radionuclides diffuse to an overlying aquifer; in Scenario 2, there is water flow from an aquifer below the repository, which carries the radionuclides to an overlying aquifer. The model pathways to man include

direct access to the water in the aquifer and via two types of river fed by the aquifer, one simulating a river like the Colorado River and the other simulating a "midwestern river." In both the river models, radionuclides reach man through direct ingestion of river water and through crops, cattle, and milk. The pathways involving direct access to the aquifer results in a "maximum" individual dose commitment that is believed unrealistic and serves principally as an upper limit value. For the river pathways, "maximum" and "reasonable" population dose commitments are calculated. The former is analogous to the "maximum" individual dose and again is an upper limit; the latter is an attempt to represent realistic conditions whereby radionuclides will be introduced into a population served by a particular body of water.

The radionuclides used in the dose calculations are those that persist or grow over long time periods and that contribute significantly to the total dose commitment. The times for the dose calculations are those times in the future at which the concentrations of the radionuclides reach maximum values.

Thus, the dose calculations are made for present-day people, with present metabolism and present modes of access to the radionuclides, using those radionuclides that would be present in the future, at their maximum concentrations, if they were indeed to move from the repository to a point accessible to such people. The calculations must, therefore, be considered only as an illustrative exercise, based on a specific set of assumptions, which cannot in any way be considered as predictive of actual occurrences in the future. Finally, all the dose commitment values that are presented can be considered to represent conservative values, partly because water has been assumed to enter the repository partly because of the use of the linear-nonthreshold dose-response model, and partly because of the transport and pathway assumptions. The expected dose commitments in a real situation would almost certainly be lower than those predicted by these illustrative models.

C-5.1.3 Conditions of Exposure and Persons or Populations at Risk

C-5.1.3.1 Usefulness of Modeling Short-Term Transport and Health Effects

Computerized simulation of the dynamics of environmental transport and redistribution of radioactivity has been most useful and valid in the studies of potential impacts of specific facilities at specific locations. In these situations, the existing environment can be investigated and substantially, though not completely, quantified for use as input to a computer model.

The model can be used to evaluate critical pathways and thereby identify the most appropriate locations for subsequent sampling and analysis. The model can also be verified after its use by comparing measured concentrations with those predicted by the model.

The extensive computer modeling of health effects resulting from the release of radionuclides to the environment, as well as of the potential mechanisms by which such releases might occur, has been valuable in pointing out areas that had been overlooked in previous evaluations of the long-term impact of nuclear energy. For example, the potential health impacts of nuclides such as carbon-14, technetium-99, iodine-129, and neptunium-237 were identified primarily as a result of computer modeling.

In nonspecific studies of facility types or typical operations, however, computerized simulations are of limited value simply because they are so critically dependent upon the assumptions of environmental conditions that are used as input data. Acceptance or rejection of the results of such modeling will depend upon whether one accepts or rejects the assumptions used as input to the model.

C-5.1.3.2 Problems in Modeling Long-Term Transport and Health Effects

The validity of detailed models for projecting transport and health effects many millenia into the future is certainly questionable. Relevant demographic and socio-economic factors can be hypothesized for such distant times, but cannot be forecast. The discussions in the following sub-sections serve to illustrate this fact.

An extensive review of 83 documented computer codes for assessing environmental transport and radiation doses from radioactivity released to the environment was published recently.⁽⁴⁹⁾ The codes evaluated exhibited a rather wide range of sophistication and capabilities. The environmental transport models and supporting data represented by these computer codes are generally similar in two respects: most of them are designed to evaluate releases from point sources (i.e., specific surface facilities) directly into the atmosphere or into a body of surface water, and most consider only the first cycle of radioactive materials through environmental pathways. No documented computer codes were found that were designed to handle releases from dispersed underground sources or to consider recycling of extremely long-lived radionuclides in the biosphere.

C-5.1.3.2.1 Population Distributions

Geographic distributions of populations tend to change as environmental conditions and resource values change. For example, anyone making predictions of present population distributions based on conditions existing more than 500 years ago would not have extrapolated beyond his own country or at least his own continent. As recently as 100 years ago, few people could have foreseen the city of Phoenix, Arizona. Even Las Vegas, Nevada, as it exists today, would have been considered unlikely as recently as 50 years ago, before the construction of Hoover Dam. In general, the fastest-growing cities in the United States during recent decades either did not exist or were inconsequential a century ago.

At the present time, the most rapid growth in the United States is occurring in the southwestern ("sunbelt") regions. The principal limitation to growth in those areas is the supply of water. However, some of those same areas may be considered for location of large Federal repositories for radioactive waste. If a practical system were devised to divert water from the northwestern region of the country, or from the Hudson Bay area of Canada, to the southwestern portion of the United States, it is entirely conceivable that extremely large metropolitan areas would develop in the Southwest. At the present time, there is no way of predicting with a high degree of assurance the population growth or distribution for long periods into the future for those areas in which HLW repositories may be considered.

C-5.1.3.2.2 Agricultural Distribution

The geographic distribution of agriculture in the United States has changed significantly during the nation's history. Two hundred years ago, few would have predicted that the Imperial Valley of California would be a major agricultural center. Also, few at that time would have predicted that the plains of Kansas, Nebraska, the Dakotas, etc., would be agriculturally important to the United States and to the world. Just as the Imperial Valley in California was changed by large-scale irrigation practices, so also could large portions of the southwest be changed by the importation of large quantities of water from more northern regions of the United States, or possibly Canada. As with population distribution, there is no assured way of predicting what the agricultural distribution might be around an HLW repository at times far into the future.

C-5.1.3.2.3 Agricultural Technology

Basic agricultural technology may change as a result of efforts to feed larger numbers of people on smaller parcels of land. This would be particularly true if continued population growth caused more agricultural land to be taken out of use for other purposes, such as residential, commercial, or industrial uses. During the past century, especially during the past few decades, a number of significant changes have been taking place in agricultural technology, particularly in the areas of increased automation and artificial enhancement of food production.

In recent years, many revolutionary procedures have been developed for producing the world's food supplies with smaller commitments of land and in more stringently controlled environments. Thus, it is entirely plausible that within a few decades, or at most within a few centuries, present-day concepts of transport of contaminants through food chains will be obsolete and irrelevant. It is possible that food production will be very closely controlled and managed in order to maximize its quantity and quality. At that time, the uncontrolled outdoor environment may have little or no interaction with the controlled agricultural environment.

Dietary and nutritional practices will undoubtedly change as the result of changes in agricultural technology and distribution networks for food products. In the past few decades, food chains have become more truly worldwide, as opposed to being localized within agricultural communities. At the same time, however, there has been a massive increase in the control of food ingredients and distribution.

C-5.1.3.2.4 Resource Conservation

As natural resources become more expensive to obtain, increased emphasis is placed on recycling. Not only mineral resources, but also organic materials are being recycled; a common example is that of wood and fiber products, such as paper.

Another resource that is being conserved and "recycled" in some parts of the country, out of sheer necessity, is water. As a nation we are rapidly approaching "limited pollution" releases with the long-range objective of "zero release." Admittedly, it will be a long time before we arrive at "zero release," but as we approach this condition, the importance of contamination from outside sources will diminish.

C-5.1.3.2.5 Medical Technology

One of the most important factors affecting the significance of radiation dose to the human race in the centuries and millenia to come is the effect of progress in medical technology. It is a matter of record that the progress achieved during the past several decades has already lengthened human life expectancy and increased the average age of the population. The success of medical science has also been felt in the increased numbers of people in the procreative age group who have survived what would once have been a fatal disease and who are now capable of passing that hereditary defect on to their offspring. It is entirely conceivable that, within the next century, the somatic diseases caused by radiation will be treated successfully, so the percentage of people who have sustained some physiological, and perhaps even genetic, damage and who are still able to produce children will increase. If, in fact, the adverse health effects of radiation prove to be either preventable or curable by advances in medical science, the impact of the radiation doses resulting from long-term storage of HLW may be substantially altered. Although none of these possible eventualities would affect the radiation doses received by future populations, the consequences of those doses could obviously be very different.

C-5.1.4 Implications of the Use of a Linear-Nonthreshold Dose-Response

The linear-nonthreshold hypothesis is defined as: "the assumption that a dose-effect curve derived from data in the high dose and high dose-rate ranges may be extrapolated through the low dose and low dose-rate range to zero, implying that, theoretically, any amount of radiation will cause some damage."⁽⁵⁰⁾ To permit simplified calculations of the health impacts of small individual radiation doses delivered to large populations, it is customary to assume that every increment of dose represents the same impact, regardless of when, where, how, and to whom it was delivered. At doses or dose-rates to individuals that are small fractions of those due to natural sources, the question of the validity of this assumption is unresolved.

Inasmuch as a straight-line extrapolation from the high-dose region downward through the origin would overestimate response effects in the low-dose

region, compared with a curvilinear form, it is generally agreed that the "linear-nonthreshold" model provides a "conservative" basis for public policy making. In addition, since this model, like all linear models, is intrinsically easy to understand and apply, there is a practical reason for preferring it to a more complicated one. The BEIR Committee, (50) therefore, recommended its adoption, in spite of the fact that experimental evidence indicated that some non-linear function would more accurately represent the true dose-response relationship for human health effects. The Committee justified its recommendation on the basis that a "non-linear hypothesis for estimating risks in support of public policy would be impractical in the present state of knowledge*, and partly because "the linear hypothesis, which allows the mean tissue dose to be used as the appropriate measure of radiation exposure, provides the only workable* approach to numerical estimations of the risk in a population."

A further advantage of using the linear-nonthreshold model arises in calculating limits of population radiation doses for circumstances where population distributions and radionuclide concentrations may not be well defined.

C-5.2 BASIS FOR DOSE-TO-MAN CALCULATIONS

C-5.2.1 Introduction

The preceding section has identified and emphasized the uncertainties and inadequacies inherent in modeling future events and consequences that require detailed knowledge of the biosphere and of human living conditions in times far into the future. In spite of such limitations, there is a distinct need for making reasonable projections related to the effects of current actions on future generations. In order to make such reasoned, though limited, projections, one must rely on making explicit assumptions and proceeding thereafter to evaluate consequences of an action or actions based on those assumptions.

For this report the dose-to-man consequences of HLW emplacement in deep geologic formations have been exemplified by stipulating a "most likely" release mechanism and by stipulating conditions whereby "maximum individual dose" and "reasonable population dose" would occur. The general

*Emphasis added

mechanisms and assumptions have been described briefly in Section 5.1.2.2. The most likely release mechanism is contact with, and subsequent leaching by, groundwater. Dose-to-man calculations have been made for two sub-categories of this release mechanism: transport by passive aqueous diffusion and by active flow of radionuclides in potable groundwater. In the following sections, the assumptions made relative to the two release mechanisms (scenarios), the three pathways, and the three categories of dose calculations are discussed in greater detail.

C-5.2.2 Repository Release Scenarios

The two release scenarios used for this task are those termed "normal" as specified in Section C-4.4. Both scenarios consider groundwater in contact with the repository waste materials, leaching of the waste, and subsequent transport through the geosphere to the biosphere. Leaching by groundwater appears to be the most likely failure mode for a waste repository. In fact, some workers have concluded that eventual contact between groundwater and the repository will occur.⁽³⁷⁾ The following sections describe the two groundwater release scenarios upon which dose-to-man calculations are based. Table C-17 presents physical data and calculated 50-year dose-commitment factors for a number of long-lived radionuclides that will be present in the waste.

C-5.2.2.1 Diffusion to a Potable Aquifer (Scenario 1)

In this release scenario, radionuclides are transported from the waste repository by way of diffusion in the water through the repository horizon, from which some of them gain access to an over-lying potable aquifer (See Figure C-15). Table C-18 shows the concentrations of important radionuclides in the aquifer directly above the repository at time periods of 300, 1000, and 30,000 years (times refer to age of waste following emplacement). The assumptions on which these concentrations are predicted include the effects of sorption and dilution in the aquifer but not sorption in the repository strata. After entering the aquifer, radionuclides move down the hydraulic gradient with the groundwater. Concentrations of important radionuclides (i.e., those that are predicted to appear in significant quantities) one mile down gradient from the repository area are also shown in

TABLE C-17

PHYSICAL DATA AND DOSE COMMITMENT FACTORS
FOR SIGNIFICANT RADIONUCLIDES

Nuclide	Half-Life (years)	Specific Activity (Ci/g)	Ingestion 50-Year Dose Commitment to Adult (rem/ μ Ci intake)*	
			Whole Body	Bone
C-14	5.7×10^3	4.5×10^0	5.65×10^{-4}	2.8×10^{-3}
Sr-90	2.8×10^1	1.4×10^2	1.67×10^{-1}	8.3×10^0
Tc-99	2.1×10^5	1.7×10^{-2}	4.92×10^{-5}	1.2×10^{-4}
I-129	1.7×10^7	1.6×10^{-4}	9.32×10^{-3}	$[7.5 \times 10^{0+}$ Thyroid]
Cs-137	3.0×10^1	8.8×10^1	4.32×10^{-2}	8.1×10^{-2}
Sm-151	8.7×10^1	2.8×10^1	2.84×10^{-6}	6.9×10^{-5}
Ra-226	1.6×10^3	1.0×10^0	3.11×10^1	$3.0 \times 10^{1\dagger}$
Th-230	8.0×10^4	2.0×10^{-2}	5.70×10^{-2}	2.0×10^0
U-233	1.6×10^5	9.7×10^{-3}	5.24×10^{-2}	8.6×10^{-1}
U-234	2.5×10^5	6.2×10^{-3}	5.13×10^{-2}	8.3×10^{-1}
U-235	7.1×10^8	2.2×10^{-6}	4.82×10^{-2}	7.9×10^{-1}
U-236	2.4×10^7	6.4×10^{-5}	4.92×10^{-2}	7.9×10^{-1}
U-238	4.5×10^9	3.4×10^{-7}	4.50×10^{-2}	7.6×10^{-1}
Np-237	2.1×10^6	7.3×10^{-4}	5.54×10^{-2}	1.4×10^0
Pu-238	8.6×10^1	1.8×10^1	1.72×10^{-2}	6.8×10^{-1}
Pu-239	2.4×10^4	6.3×10^{-2}	1.91×10^{-2}	7.9×10^{-1}
Pu-240	6.6×10^3	2.3×10^{-1}	1.91×10^{-2}	7.9×10^{-1}
Pu-242	3.8×10^5	3.9×10^{-3}	1.84×10^{-2}	7.3×10^{-1}
Am-241	4.6×10^2	3.3×10^0	5.42×10^{-2}	8.2×10^{-1}
Am-243	8.0×10^3	1.9×10^{-1}	5.30×10^{-2}	8.2×10^{-1}

*Taken from Table 4.3, ORNL-4992⁽⁵⁴⁾ calculated by INREM for a single intake at age 21.

⁺ 2.8×10^1 for child.

[†]From ICRP Publ. 10.

TABLE C-18

RADIONUCLIDE CONCENTRATIONS FOLLOWING DIFFUSION
INTO A POTABLE AQUIFER (SCENARIO 1)*

Nuclide	Aquifer Entry			One Mile Below Repository			
	300 Years	1000 Years	30,000 Years	300 Years	1000 Years	30,000 Years	100,000 Years
C-14	4.3x10 ⁻¹⁹	7.6x10 ⁻¹⁶	††	1.3x10 ⁻²⁵	1.4x10 ⁻¹⁷	††	††
Sr-90	1.2x10 ⁻¹⁹	3.3x10 ⁻²⁴	††	-	-	††	††
Tc-99R [†]	2.4x10 ⁻²⁰	1.4x10 ⁻¹⁶	1.7x10 ⁻¹¹	-	-	-	††
Tc-99N [†]	1.6x10 ⁻¹³	2.4x10 ⁻¹⁰	**	3.1x10 ⁻¹³	1.7x10 ⁻¹⁰	**	**
I-129	4.4x10 ⁻¹⁴	6.5x10 ⁻¹¹	**	8.6x10 ⁻¹⁴	4.7x10 ⁻¹¹	**	**
Cs-137	7.2x10 ⁻²⁰	8.7x10 ⁻²⁴	††	-	-	††	††
Sm-151	1.4x10 ⁻¹⁸	1.4x10 ⁻¹⁷	††	-	-	††	††
U-235	2.7x10 ⁻¹⁵	8.1x10 ⁻¹²	5.5x10 ⁻⁶	-	-	-	††
U-236	1.4x10 ⁻¹⁵	4.4x10 ⁻¹²	3.1x10 ⁻⁶	-	1.0x10 ⁻²³	2.0x10 ⁻⁶	††
Np-237	4.6x10 ⁻¹⁴	1.1x10 ⁻¹⁰	2.3x10 ⁻⁶	-	-	-	††
Pu-239	2.5x10 ⁻¹⁵	7.1x10 ⁻¹²	1.5x10 ⁻⁶	-	-	-	††
Pu-240	1.0x10 ⁻¹⁵	2.6x10 ⁻¹²	6.3x10 ⁻⁸	-	-	-	††
Am-243	4.1x10 ⁻¹⁷	1.0x10 ⁻¹³	3.4x10 ⁻⁹	-	-	-	††
Ra-226	8.7x10 ⁻²¹	6.4x10 ⁻¹⁶	2.8x10 ⁻⁹	-	-	5.5x10 ⁻¹⁶	1.2x10 ⁻¹³

*All values are representative of the throwaway cycle and are in units of g/L.

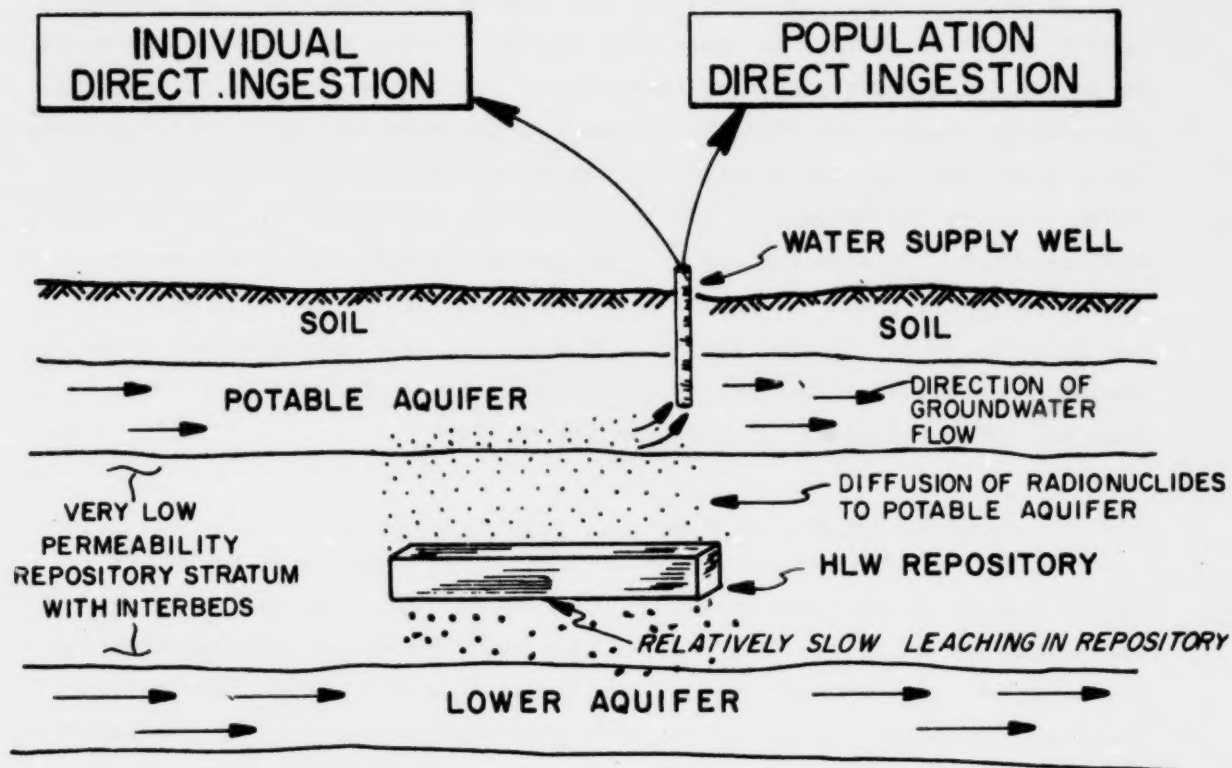
**Concentrations at these time intervals are less than concentrations at 1000 years.

†Tc-99R represents an assumed adsorptive ($R_d=3300$) form of Tc-99.

††Tc-99N represents a non-adsorptive ($R_d=1$) form of Tc-99.

Concentrations at these times are not available.

- Dashes indicate insignificant concentrations (less than one atom per liter).



Source: D'Appolonia Consulting Engineers, Inc., for Arthur D. Little, Inc.

FIGURE C-15 SCHEMATIC ILLUSTRATION OF MAXIMUM INDIVIDUAL AND UPPER LIMIT POPULATION DOSE BY DIFFUSION TO A POTABLE AQUIFER (SCENARIO 1)

Table C-18 for time periods of 300, 1000, 30,000 and 100,000 years following waste emplacement.

C-5.2.2.2 Flow to a Potable Aquifer (Scenario 2)

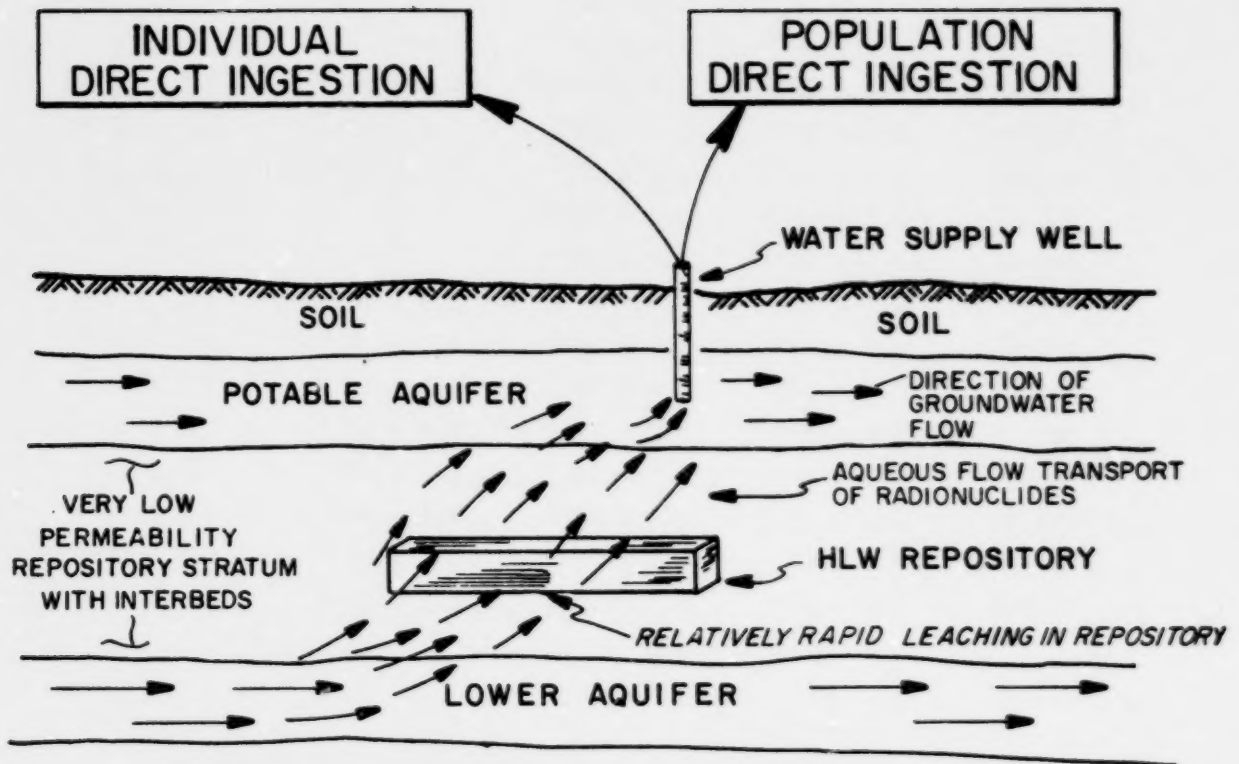
In this scenario, radionuclides are transported from the waste repository by groundwater from an underlying aquifer flowing through the repository to an overlying potable aquifer (See Figure C-16). When the nuclide-bearing groundwater reaches the aquifer, movement is downgradient within that aquifer. Table C-19 presents concentrations of important radionuclides in the aquifer directly above the repository and in the aquifer one mile down gradient from the repository. Concentrations at time periods of 300, 500, 6000, and 30,000 years are shown for the area overlying the repository site and for 300, 500, 6000, 30,000, and 100,000 years one mile down gradient from the repository. These concentration estimates are based on assumptions that include sorption and dilution in the aquifer.

C-5.2.3 Pathway Models

The intake of radionuclides by people is postulated to occur not by direct access to the repository but by access through some pathway to the aforementioned aquifer. Three routes have been chosen for consideration, one of which (direct access to the contaminated aquifer) would require concurrence of several specific conditions or events and may therefore be rather unlikely, while the other two (river models) are intuitively more plausible and realistic. In each of the three cases, contamination of the aquifer was assumed to occur by either the Scenario 1 or Scenario 2 mechanisms. Information important to the use of all models for calculating dose commitments is listed in Table C-20.

C-5.2.3.1 Direct Access to Aquifer

In this case the assumption is made that water is obtained from a well that has been drilled into the aquifer, i.e., the resulting pathway is that of direct ingestion of the contaminated water continuously for a period of one year. This is a worst-case situation, most likely to be prevented either by appropriate security or by the remoteness and lack of



Source: D'Appolonia Consulting Engineers, Inc., for Arthur D. Little, Inc.

FIGURE C-16 SCHEMATIC ILLUSTRATION OF MAXIMUM INDIVIDUAL AND UPPER LIMIT POPULATION DOSE BY AQUEOUS FLOW TRANSPORT TO A POTABLE AQUIFER (SCENARIO 2)

TABLE C-19

Radionuclide Concentrations* in Aquifer at Aquifer Entry (Scenario 2)

<u>Nuclide</u>	<u>300 years</u>	<u>500 years</u>	<u>6000 years</u>	<u>30,000 years</u>
C-14	1.7×10^{-9}	1.3×10^{-9}	††	††
Sr-90	2.6×10^{-8}	5.3×10^{-10}	††	††
Tc-99R [†]	6.7×10^{-11}	2.4×10^{-10}	4.5×10^{-10}	2.1×10^{-10}
Tc-99N [†]	5.1×10^{-4}	3.2×10^{-4}	2.3×10^{-5}	4.2×10^{-6}
I-129	1.4×10^{-4}	8.7×10^{-5}	6.4×10^{-6}	1.3×10^{-6}
Cs-137	1.2×10^{-8}	4.8×10^{-10}	††	††
Sm-151	2.0×10^{-8}	1.6×10^{-8}	††	††
U-233	7.5×10^{-11}	4.2×10^{-10}	1.9×10^{-8}	3.6×10^{-8}
U-234	2.8×10^{-7}	1.0×10^{-6}	4.1×10^{-6}	3.4×10^{-6}
U-236	3.9×10^{-6}	1.4×10^{-5}	7.8×10^{-5}	7.3×10^{-5}
Np-237	1.0×10^{-4}	3.0×10^{-4}	2.4×10^{-5}	1.1×10^{-5}
Pu-240	2.9×10^{-6}	1.0×10^{-5}	1.2×10^{-5}	2.8×10^{-6}
Pu-242	5.9×10^{-7}	2.2×10^{-6}	8.1×10^{-6}	5.5×10^{-6}
Am-243	1.2×10^{-7}	4.1×10^{-7}	5.5×10^{-7}	1.4×10^{-7}
Ra-226	8.6×10^{-12}	8.4×10^{-11}	1.3×10^{-8}	6.6×10^{-8}

* All values are representative of the throwaway cycle and are in units of g/l.

[†] Tc-99R represents an assumed adsorptive ($R_d=3300$) form of Tc-99.
Tc-99N represents a non-adsorptive ($R_d=1$) form of Tc-99.

†† Concentrations at these times are not available.

TABLE C-19 (Continued)

Nuclide Concentration in Aquifer One Mile Beyond Repository* (Scenario 2)

<u>Nuclide</u>	<u>300 years</u>	<u>500 years</u>	<u>6000 years</u>	<u>30,000 years</u>	<u>100,000 years</u>
C-14	-	1.2×10^{-14}	††	††	††
Sr-90	-	-	††	††	††
Tc-99R [†]	-	-	-	3.1×10^{-19}	8.5×10^{-14}
Tc-99N [†]	1.3×10^{-4}	3.6×10^{-4}	2.3×10^{-5}	4.2×10^{-6}	††
I-129	3.7×10^{-5}	1.0×10^{-4}	6.5×10^{-6}	1.3×10^{-6}	††
Cs-137	-	-	††	††	††
Sm-151	-	-	††	††	††
U-233	-	1.0×10^{-20}	1.8×10^{-9}	2.7×10^{-9}	2.7×10^{-9}
U-234	-	-	-	-	-
U-236	-	-	-	-	9.1×10^{-21}
Np-237	-	2.2×10^{-14}	7.4×10^{-5}	1.8×10^{-5}	1.1×10^{-9}
Pu-240	-	-	-	-	5.6×10^{-23}
Pu-242	-	-	-	-	1.2×10^{-19}
Am-243	-	-	-	-	2.4×10^{-19}
Ra-226	-	-	4.4×10^{-18}	8.0×10^{-14}	1.6×10^{-12}

* All values are representative of the throwaway cycle and are in units of g/l.

[†] Tc-99R represents an assumed adsorptive ($R_d=3300$) form of Tc-99.
Tc-99N represents a non-adsorptive ($R_d=1$) form of Tc-99.

^{††} Concentrations at these times are not calculated.

Note: Dashes indicate the concentration of the particular nuclide is less than one gram per liter.

TABLE C-20

**ENVIRONMENTAL AND BIOLOGICAL TRANSFER
COEFFICIENTS FOR DOSE COMMITMENT CALCULATIONS**

1. Intakes:**A. Drinking Water***

1. Maximum Adult - 730 l/year
2. Average Adult - 370 l/year
3. Maximum Child - 510 l/year
4. Average Child - 260 l/year
5. Beef Cattle - 50 l/day
6. Milk Cows - 60 l/day

B. Feed Crops*

1. Beef Cattle - 50 kg/day
2. Milk Cows - 50 kg/day

2. Usage Rates:

- A. River Model A, Upper Basin Irrigation - $0.23 \text{ l/m}^2\text{-hr}$
 River Model A, Lower Basin Irrigation - $0.11 \text{ l/m}^2\text{-hr}$
- B. River Model B, Total Basin Irrigation - $0.011 \text{ l/m}^2\text{-hr}^+$

3. Stable Element Transfer Coefficients:*

Element	B_v^+ (veg/soil)	F_b^+ (Beef: days/kg)	F_m^+ (Milk: days/l)
Tc	2.5×10^{-1}	1.0×10^{-3}	2.5×10^{-2}
I	2.0×10^{-2}	4.0×10^{-2}	6.9×10^{-3}
Ra	3.1×10^{-4}	3.4×10^{-2}	1.5×10^{-2}
Np	2.5×10^{-3}	2.0×10^{-4}	5.0×10^{-6}

* ORNL-4992. (54)

⁺ EPA-520/4-73-002. (55)

⁺ See Appendix C-V for explanation of transfer coefficients.

usefulness of the repository area. It provides, for reference purposes, an extreme case of human exposure to a contaminated aquifer, and produces an "unrealistically" high radiation dose commitment to affected individuals.

C-5.2.3.2 River Model "A"

This river model is constructed using data typical of the Colorado River mainstream. Water from the aquifer is assumed to enter the river one mile beyond the repository. Main water diversions (for population use) along the river are taken at five major locations.

The river model has an upper and lower basin; the upper basin has more rapid dilution via tributaries, and the lower basin has a relatively constant flow and dilution factor. For this reason, two diversions were used to represent use in the upper basin and one "total diversion" to represent use in the lower basin. A single point diversion in the lower basin (at the river mouth) was used because the dilution factor throughout the lower basin is approximately constant. Other upper-basin characteristics are high agricultural use and relatively small populations ingesting water.

Dilution factors applicable at each diversion were calculated using a general materials balance. Use along the river has been taken either from existing Colorado River data, as stated in Table C-21, or calculated from standard use and yield values.⁽⁵¹⁾

C-5.2.3.3 River Model "B"

This model is for a hypothetical river having a population and use distribution proportional to the river flow (i.e., proportional to the dilution factor). Water from the aquifer is assumed to enter the river one mile beyond the repository. The model was previously used for a purpose similar to its application in this assessment.⁽⁵²⁾ This hypothetical river can be considered more typical of a midwestern river than River Model "A" in that it is more heavily used for drinking water and less used for agricultural supply. Where available data are insufficient, the same assumptions and water use calculations made for River Model "A" are applied in River Model "B". Information important in using River Model "B" for calculating a reasonable population dose is listed in Table C-22.

TABLE C-21
INFORMATION FOR RIVER MODEL "A"^a
FOR REASONABLE POPULATION DOSE CALCULATIONS

Distance to Diversion (miles)	River Flow (cfs)	Dilution Factor	Diversion (cfs)	Population Drinking Water (persons)	Total Irrigation Acreage (acres)	Sprinkler Irrigation Acreage (acres)	Total Crop Yield (kg/yr)	Beef [†] Feed (kg/yr)	Dairy [‡] Cow Feed (kg/yr)	Beef [§] Yield (kg/yr)	Milk [¶] Yield (L/yr)	Garden Vegetable Yield (kg/yr)
<u>Upper Basin</u>												
0	2,400	1.0	-	-	-	-	-	-	-	-	-	-
50	4,000	1.7	1,600	1×10^4	1.5×10^5	1.5×10^4	4.5×10^8	3.9×10^8	4.0×10^7	3.9×10^6	3.2×10^7	2.0×10^7
100	6,700	4.5	700	1×10^5	6.6×10^4	6.6×10^3	2.0×10^8	1.8×10^8	2.0×10^7	1.8×10^6	1.6×10^7	-
<u>Lower Basin</u>												
550	13,700	10.4	1,500	4×10^5	-	-	-	-	-	-	-	-
700	12,300	10.5	1,600	5×10^6	-	-	-	-	-	-	-	-
800	11,500	11.3	4,000	1×10^6	-	-	-	-	-	-	-	-
		11.3	7,100	$5.4 \times 10^{6**}$	1.5×10^6	1.5×10^5	1.2×10^{10}	-	-	-	-	$1.2 \times 10^{10++}$

^a Assembled using Colorado River mainstream data.

[†] Assumes 10 percent to be contaminated by sprinkler irrigation.

[‡] Assumes 90 percent of animal feed is for beef cattle, 10 percent is for dairy cows.

[§] Assumes 1 kg beef per 100 kg feed.

[¶] Assumes 80 L milk per 100 kg feed.

^{**} Assumes all crop yield is for human ingestion.

⁺⁺ Used one "total" diversion for the lower basin.

Note: Dashes indicate instances where the information was not used for calculational purposes and consequently is not presented.

TABLE C-22
INFORMATION FOR RIVER MODEL "B"*
FOR REASONABLE POPULATION DOSE CALCULATIONS

Distance to Diversion (miles)	River Flow (cfs)	Dilution Factor	Population Drinking Water (persons)	Total Crop Yield (kg/yr)	Beef Feed (kg/yr)	Dairy Cow Feed (kg/yr)	Beef† Yield (kg/yr)	Milk† Yield (l/yr)	Garden Vegetable Yield (kg/yr)
0	200	1.0	-	-	-	-	-	-	-
15	220	1.1	4.3 x 10 ³	2.0 x 10 ⁵	3.0 x 10 ⁴	6.2 x 10 ⁴	3.0 x 10 ²	5.0 x 10 ²	1.1 x 10 ⁵
45	360	1.8	7.0 x 10 ³	3.2 x 10 ⁵	4.7 x 10 ⁴	9.8 x 10 ⁴	4.7 x 10 ²	7.8 x 10 ⁴	1.8 x 10 ⁵
80	700	3.5	1.8 x 10 ⁴	8.4 x 10 ⁵	1.2 x 10 ⁵	2.5 x 10 ⁵	1.2 x 10 ³	2.0 x 10 ³	4.7 x 10 ⁵
150	2,000	9.9	1.3 x 10 ⁵	6.0 x 10 ⁶	8.6 x 10 ⁵	1.8 x 10 ⁶	8.6 x 10 ³	1.4 x 10 ⁶	3.3 x 10 ⁶
250	5,200	26	3.3 x 10 ⁵	1.5 x 10 ⁷	2.2 x 10 ⁶	4.6 x 10 ⁶	2.2 x 10 ⁴	3.7 x 10 ⁶	8.6 x 10 ⁶
350	10,000	50	6.3 x 10 ⁵	2.9 x 10 ⁷	4.2 x 10 ⁶	8.7 x 10 ⁶	4.2 x 10 ⁴	7.0 x 10 ⁶	1.6 x 10 ⁷
450	16,000	81	1.1 x 10 ⁶	4.8 x 10 ⁷	6.9 x 10 ⁶	1.4 x 10 ⁷	6.9 x 10 ⁴	1.1 x 10 ⁷	2.7 x 10 ⁷

* Assembled using river in "GESMO" (NUREG-0002, Vol. 3). (52)

† Assumes 1 kg beef per 100 kg feed.

‡ Assumes 80 l milk per 100 kg feed.

Note: Dashes indicate instances where the information was not used for calculational purposes and consequently is not presented.

C-5.2.3.4 Accumulation of Radionuclides in Surface Soil

Continual irrigation with contaminated water produces a long-term buildup of contaminants in surface soil. At the same time, contaminants are continually removed by erosion, by uptake into plants, or by leaching below the root zones of plants (effective removal from food chains). There are few data available to indicate the natural residence time of various elements in tillable soils. For this study, a soil exposure time of 100 years was used to allow for reasonable buildup of radioactive contaminants through time. For calculational purposes the midpoint of this time, 50 years, is used as an average value for transfer coefficients from soil to plants. This is believed to be a conservative assumption, since it is unlikely that most nuclides will accumulate without dispersion for that period of time.

C-5.2.4. Exposure Routes

For the postulated "most likely" mechanisms for release of radionuclides from an HLW repository (i.e., leaching and transport by groundwater) the "maximum" individual and population doses can be approximated without detailed pathway modeling. This results because, for most of the radionuclides, direct exposure routes lead to much higher radiation doses than do the indirect exposure routes. The indirect routes, however, are the more likely ones whereby man could be exposed to HLW radionuclides.

One possible exception to the direct route being more important is iodine-129 for which the radiation dose via the milk-to-human pathway can be approximately 100 times greater (per unit of liquid ingested) than the dose from direct ingestion of the radioiodine in drinking water. However, such extreme differences between doses delivered by direct and indirect pathways occur only with the simultaneous existence of two or more rather unlikely maximizing circumstances. For example, in order to have an annual intake approximately 100 times greater than would be achieved by direct ingestion of the contaminated water, a child would have to drink nothing but milk from cows whose only source of feed was vegetation grown on land totally irrigated with contaminated water at an irrigation rate of one meter/

year and with a retention of radiiodine on the edible portions of the plant of 25% of the total amount sprinkled. In similar fashion, a child obtaining all its vegetable food products from land irrigated with contaminated water under the same circumstances as described above would also receive a radiation dose approximately 100 times greater than that delivered by direct ingestion of the contaminated water. Such circumstances are not considered realistic for purposes of dose estimations.

Models, assumptions, and data have been prepared or collected by the U.S. Nuclear Regulatory Commission (NRC) for the purpose of calculating radiation doses to individuals and populations.⁽⁵¹⁾ This information was evaluated and was determined to be adequate, reasonable, and realistic for the purposes of this study. Consequently the NRC models, assumptions and data have been utilized for dose calculations; these will be addressed more explicitly in the subsequent sections.

The conversion factors for the radiation dose to an individual, per unit of radioactivity ingested or inhaled, are based on the Oak Ridge computer dose program INREM as described in ORNL-5003⁽⁵³⁾ and as tabulated in ORNL-4992.⁽⁵⁴⁾ The dose conversion factors tabulated in ORNL-4992 are 50-year dose commitments to an individual at age 21. These conditions are felt to be a suitable approximation for an average population when all other uncertainties in the assumptions are considered.

Generalized environmental transfer and bioaccumulation coefficients for a rather complete range of radionuclides have been used for environmental modeling as shown in Table C-22 for River Model "B".⁽⁵¹⁾ These coefficients are appropriate for use where site-specific data are not available; this certainly is the case for considerations relative to HLW repositories in geologic formations.

C-5.2.4.1 "Maximum" Individual Radiation Dose

The "maximum" individual dose is calculated on the assumption that an individual is present at the point of maximum environmental concentration for a specified radionuclide and at the point in time for which that nuclide reaches the maximum concentration. It is assumed, however, that the individual ingests the contaminated material at an estimated maximum

rate (730 liters per year) and is an individual of average age in the population. In this respect, the calculation should not be considered the "maximum conceivable individual dose" but is more appropriately thought of as a "maximum dose to the typical individual."

The "maximum" individual dose calculations are based on the assumption that an individual would obtain all drinking water from a well penetrating the contaminated aquifer at the point of highest concentration in the vicinity of the waste repository (see Figures C-15 and C-16). Thus, the maximum individual dose in this case is calculated by assuming that it results from continuous consumption of water by a 21-year-old individual for a period of one year, that the water is contaminated to the maximum level in the aquifer, and that it produces a 50-year dose commitment during the lifetime of the individual. For purposes of calculating a 50-year dose commitment, an ingestion period of one year is considered to be the same as a single instantaneous ingestion. The location of the well in the aquifer is assumed to be directly above the repository where the radionuclides in the aquifer are at the maximum concentration following initial dilution. The times used for estimating the radionuclide concentrations in each scenario are those that produce the maximum radionuclide dose commitments for fission products and actinides, respectively.

C-5.2.4.2 "Maximum" Population Radiation Dose

A "maximum" population dose is closely related to the "maximum" individual dose. In most cases of release of radionuclides into the environment, the assumed conditions of release generate certain limitations to the total population radiation dose that can be delivered, at least by direct exposure pathways. Analogous to the "maximum" individual dose, the "maximum" population dose is designed to represent an upper limit of radiation dose to a population present at the time and location for which the maximum impact might occur from a release of radionuclides from a waste repository. However, the population density and the environmental transfer coefficients are assumed to be approximately average for communities that might be exposed, rather than the extremes that might produce the "maximum conceivable" population dose. In the case of a contaminated water body such as an aquifer, a surface reservoir, or a surface stream,

the "maximum" population radiation dose is equal to the "maximum" individual radiation dose (i.e., the radiation dose to an individual drinking a maximum quantity [730 l/yr] of the contaminated water exclusively) multiplied by the population (250 persons) that could be supplied by the water body (see Figures C-15 and C-16), and reduced by a factor of two to allow for a lower annual water intake of 370 l/person. The upper limit of population that can be supplied by a particular water body was estimated on the basis of current per-capita domestic water consumption and the total outflow volume of the hypothetical aquifer.

C-5.2.4.3 "Reasonable" Population Radiation Dose

"Maximum" radiation doses to individuals and populations represent upper bounds for the analysis of health consequences. However, they do not represent what might cautiously be called "most reasonable estimates" of radiation doses that might be expected from a geologic HLW repository. It is highly unlikely that any individual or population group would be present at precisely the point of maximum environmental concentration in the event of release of radionuclides from a repository. It seems more realistic to postulate that populations would be exposed at some distance from the repository and through indirect pathways rather than by the most direct pathways. To develop such realistic population radiation dose estimates for unspecified repository locations and indeterminate times into the future, it has been necessary to make many simplifying assumptions.

"Reasonable" population radiation doses are calculated on the basis of the few radionuclides that actually reach the biosphere under the assumed exposure mode, i.e., by discharge of a contaminated underground aquifer into a surface stream. These few critical radionuclides are assumed to follow typical pathways and reach typical populations (based on current societal and environmental conditions) during the course of their migration downstream toward the ocean. The typical environmental pathways are simplified, primarily to emphasize the fact that the calculations should not be thought of as highly accurate but as subject to some uncertainty. Population distributions and densities, agricultural practices, etc., are assumed to be those common in much of the United

States at present. No reasons could be found for anticipating particular population densities or agricultural practices at times in the distant future.

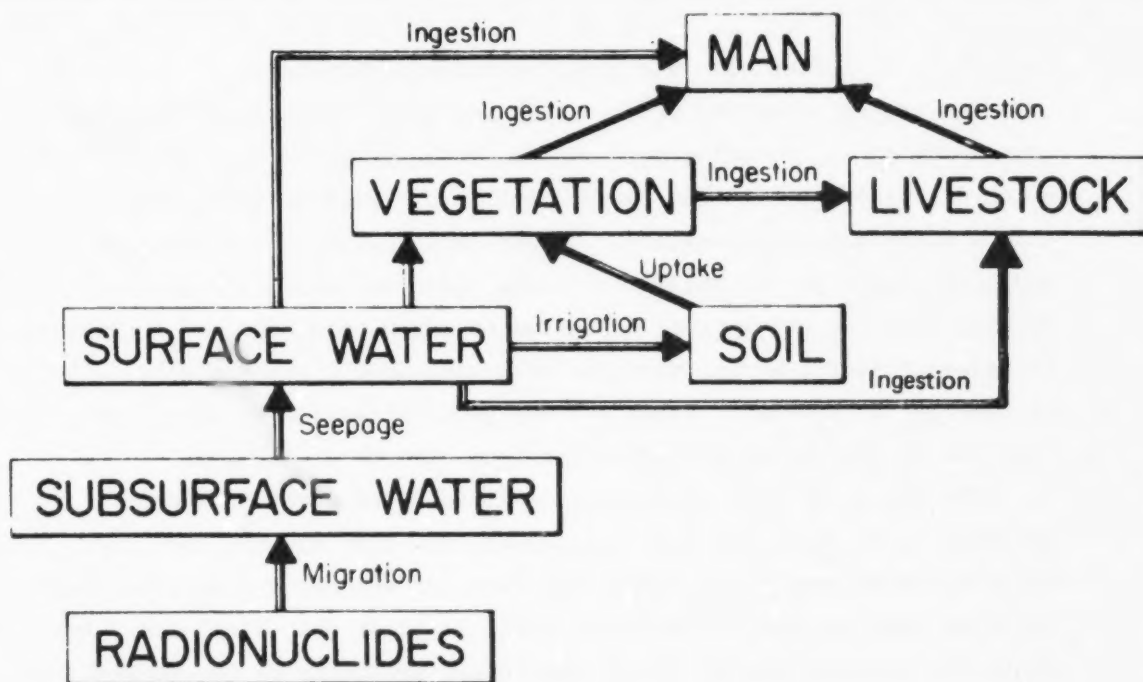
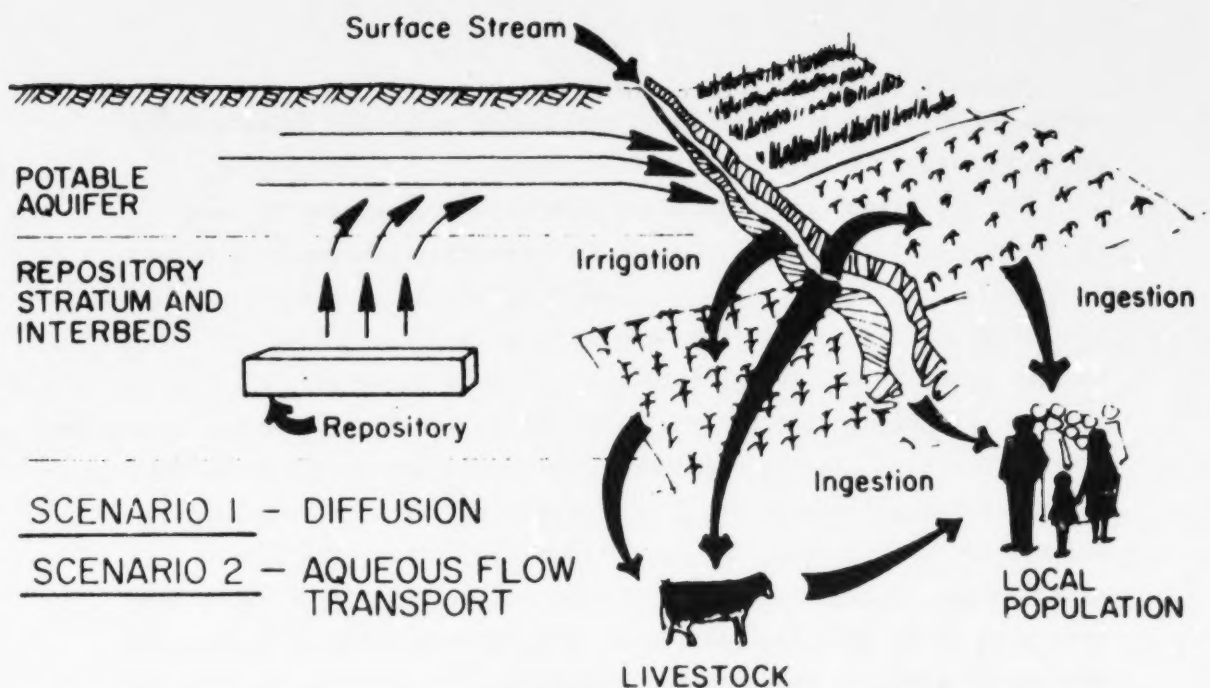
The "reasonable" population dose commitments for each scenario have been calculated on the basis of the contaminated aquifer intersecting a surface stream one mile down-gradient from the repository. Ingestion pathways used in these calculations include:

- (1) Direct ingestion of contaminated river water;
- (2) Ingestion of contaminated crops;
- (3) Ingestion of contaminated beef; and
- (4) Ingestion of contaminated milk.

These four pathways are illustrated schematically in Figure C-17 and are applied to the two river models.

The "reasonable" population dose commitment represents the 50-year dose commitment to a population resulting from one year of intake via all four pathways. Although the population is different for each pathway, the population dose commitment (person-rem) can be added for different pathways because of the assumed linear-nonthreshold dose relationship.

In almost all cases for which "reasonable" population radiation doses are calculated, as well as for some of the cases for which "maximum" population radiation doses are estimated, it was noted that size of population may not necessarily be of great importance when using the assumed linear-nonthreshold model. In general, when local meteorological or hydrological parameters are unknown, it could be assumed that radio-nuclides released to the environment would be distributed and diluted in approximate proportion to the area involved; i.e., the environmental concentration would be inversely proportional to the area over which the material was spread. On the other hand, lacking definite population distribution data, making an assumption of uniform population density would imply that the affected population would increase in proportion to the affected area. Thus, if the transfer coefficients from environmental media to human beings remain constant, the total population radiation dose would tend to remain constant, regardless of the area or the population over which a radioactive contaminant is spread. While there are exceptions to this generalization, it is nevertheless



Source: D'Appolonia Consulting Engineers, Inc., for Arthur D. Little, Inc.

FIGURE C-17 SCHEMATIC ILLUSTRATION OF "REASONABLE" POPULATION DOSE (SCENARIOS 1 & 2)

useful in arriving at upper bound population radiation doses without specific data on population distributions.

The population radiation dose from food crops grown on land contaminated with irrigation water is inherently limited by a number of factors. The total amount of land that can be irrigated and, hence, the total amount of crops that can be grown are limited by the volume of water available. If the volume of water is increased, the concentration of radionuclides in the water, and subsequently in crops, decreases proportionately, and the transfer of radionuclides to the total food chain remains essentially constant (the transfer coefficient from soil to vegetation being assumed to remain constant). Thus, the total distribution of radionuclides to the population via first-cycle food chains is inherently limited. The upper bound population dose commitment is obtained by assuming total ingestion of the leafy vegetables that can be grown regardless of the number of people consuming the foods.

C-5.2.4.4 Time Intervals for Dose Commitment Calculations

For the calculations of dose commitment, it is necessary to make some assumptions regarding duration of the exposure. It could be assumed, for instance, that future generations will be unable to detect and evaluate radioactive contaminants in water. In that case, exposures could go on indefinitely. On the other hand, it could be assumed that, with continued expansion of human knowledge and technology, the identification and elimination of environmental contaminants would be essentially assured. Thus, for any group of exposed people, the duration of the exposure could range from days to a lifetime.

The age at which an individual ingests a radioactive material determines, in part, the dose commitment from that intake. Detailed calculations of population radiation doses for assumed age distributions and life expectancies are not warranted, as previously discussed. Age 21 is the youngest age of intake that could typically be considered average for a large population and still permit a lifetime dose commitment of 50 years. By using the 50-year dose commitment for an intake at age 21 for the entire population, instead of an

age-specific calculation for an assumed population, a realistic estimate of the "reasonable" population radiation dose is attained.

All dose commitments are normalized on the basis of a one-year intake of water or food products. This duration of exposure is not unreasonable and is also convenient for changing to other time periods if so desired. Changes in the duration of exposure will produce proportionate changes in the dose commitments. However, the conclusions in Section 5.4 would not be affected even if the duration of exposure were changed by one or two orders of magnitude.

C-5.3 RESULTS OF RADIATION DOSE CALCULATIONS

C-5.3.1 Introduction

All dose calculations presented in the following sections are developed from the geologic transport conditions described in Section C-4. Since the distribution of radionuclides in the biosphere is assumed to be independent of the time or rate of release into a potable aquifer (except as time affects the repository inventory), any change in geologic assumptions would result in calculated doses that would be changed only by a linear scaling factor.

For all of the postulated scenarios and pathways, only a few radionuclides contribute essentially all of the potential radiation dose. The results of the radiation dose calculations are present for those times at which each of these "important" nuclides reaches a maximum concentration in the aquifer. However, the total radiation dose given for the whole body and for each organ at each of these times includes not only the radiation dose from the one "important" radionuclide that has the maximum concentration at a particular time, but also radiation doses from each of the other "important" radionuclides, whatever their concentrations at this time.

The tabulations shown in the following pages reveal that the "important" radionuclides from the standpoint of radiation dose commitment are limited to strontium-90, iodine-129, radium-226, neptunium-237, and plutonium-239 and -240, each of which has a maximum contribution at a different point in time. Among the nuclides that were considered as potentially important, but were subsequently shown to be of relatively minor significance (under the circumstances assumed for the purposes of

this study), are technetium-99, cesium-137, uranium-236, and americium-243.

C-5.3.2 "Maximum" Individual Radiation Dose

The "maximum" radiation doses from various radionuclides that would be delivered to an individual who obtained all drinking water (730ℓ/year) from the aquifer directly above a repository are shown in Tables C-23, C-24, and C-25. A sample calculation is included in Appendix C-IV.

C-5.3.2.1 Diffusion to a Potable Aquifer (Scenario 1)

For the Scenario 1 (diffusion) case, the "maximum" dose commitments are very small (Table C-23). The dose commitments from fission products were calculated at 300 years, a time when the concentrations of these nuclides in the aquifer are assumed to be at a maximum.

Some of the actinide elements also contribute substantially to the total dose commitment at 300 years. The "maximum" individual whole body radiation dose at 300 years is 9×10^{-9} rem, consisting principally of almost equal contributions from strontium-90 and plutonium-239 and -240. The same three radionuclides contribute most of the bone dose, calculated to be 4×10^{-7} rem. The thyroid dose resulting from iodine-129 at 300 years would be 4×10^{-8} rem.

After 30,000 years, the fission products would no longer contribute significantly to the dose commitment of the whole body or bone. The thyroid dose commitment is maximum at 1,000 years, with a value of 6×10^{-5} rem, and decreases with longer periods of time, becoming negligible with respect to the other dose commitments at 30,000 years. Scenario 1 indicates that the concentration of actinides in the aquifer would increase continuously up to a maximum at approximately 30,000 years. At that time, the "maximum" individual whole body dose commitment is calculated to be 60 rem, contributed primarily by Ra-226. The maximum bone dose commitment at that time is calculated to be 100 rem, primarily from Ra-226 and Pu-239.

C-5.3.2.2 Flow to a Potable Aquifer (Scenario 2)

For Scenario 2, understandably large values for the "maximum" individual dose commitments are obtained (Table C-24). At 300 years, the calculated "maximum" individual dose commitment is 500 rem to the whole

TABLE C-23

FIFTY-YEAR DOSE COMMITMENT CALCULATED FOR THE CASE
OF DIFFUSION INTO A POTABLE AQUIFER (SCENARIO 1)^a

	Fission Products				Radium and Actinides							
	Sr-90	Tc-99 ($R_d = 0$)	I-129	Cs-137	Ra-226	U-236	Np-237	Pu-239	Pu-240	Am-243	TOTAL ^d	
"Maximum" Individual Dose[†]												
Time = 300 years												
Whole Body (rem)	2×10^{-9}	1×10^{-10}	5×10^{-11}	2×10^{-10}	2×10^{-10}	-	1×10^{-9}	2×10^{-9}	3×10^{-9}	3×10^{-10}	9×10^{-9}	
Bone (rem)	1×10^{-7}	2×10^{-10}	-	4×10^{-10}	2×10^{-10}	-	4×10^{-8}	9×10^{-8}	1×10^{-7}	5×10^{-9}	4×10^{-7}	
Thyroid (rem)	-	-	4×10^{-8}	-	-	-	-	-	-	-	4×10^{-8}	
Time = 1,000 years												
Whole Body (rem)	-	1×10^{-7}	7×10^{-8}	-	1×10^{-5}	-	3×10^{-6}	6×10^{-6}	8×10^{-6}	7×10^{-7}	3×10^{-5}	
Bone (rem)	-	4×10^{-7}	-	-	1×10^{-5}	2×10^{-7}	8×10^{-5}	3×10^{-4}	3×10^{-4}	1×10^{-5}	7×10^{-4}	
Thyroid (rem)	-	-	6×10^{-5}	-	-	-	-	-	-	-	6×10^{-5}	
Time = 30,000 years												
Whole Body (rem)	-	-	-	-	6×10^{-1}	7×10^{-3}	7×10^{-2}	1×10^0	2×10^{-1}	2×10^{-2}	6×10^{-1}	
Bone (rem)	-	-	-	-	6×10^{-1}	1×10^{-1}	2×10^0	5×10^1	8×10^0	4×10^{-1}	1×10^2	
"Maximum" Population Dose^{**†}												
Time = 300 years												
Whole Body (person-rem)	3×10^{-7}	1×10^{-8}	7×10^{-9}	3×10^{-8}	3×10^{-8}	-	2×10^{-7}	3×10^{-7}	4×10^{-7}	4×10^{-8}	1×10^{-6}	
Bone (organ-rem)	1×10^{-5}	3×10^{-8}	-	5×10^{-8}	2×10^{-8}	-	4×10^{-6}	1×10^{-5}	2×10^{-5}	6×10^{-7}	5×10^{-5}	
Thyroid (organ-rem)	-	-	1×10^{-5}	-	-	-	-	-	-	-	1×10^{-5}	
Time = 1,000 years												
Whole Body (person-rem)	-	2×10^{-5}	9×10^{-6}	-	2×10^{-3}	-	4×10^{-4}	8×10^{-4}	1×10^{-3}	9×10^{-5}	4×10^{-3}	
Bone (organ-rem)	-	5×10^{-5}	-	-	2×10^{-3}	-	1×10^{-2}	3×10^{-2}	4×10^{-2}	1×10^{-3}	8×10^{-2}	
Thyroid (organ-rem)	-	-	7×10^{-3}	-	-	-	-	-	-	-	7×10^{-3}	
Time = 30,000 years												
Whole Body (person-rem)	-	-	-	-	8×10^3	9×10^{-1}	9×10^0	2×10^2	3×10^1	3×10^0	8×10^3	
Bone (organ-rem)	-	-	-	-	8×10^3	1×10^1	2×10^2	7×10^3	1×10^3	5×10^1	2×10^4	
"Reasonable" Population Dose^{***}												
<u>River Model "A"</u>												
Time = 1,000 years												
Thyroid (organ-rem)	-	-	8×10^{-3}	-	-	-	-	-	-	-	8×10^{-3}	
Time = 30,000 years												
Whole Body (person-rem)	-	-	-	-	-	-	3×10^0	-	-	-	3×10^0	
Time = 100,000 years												
Bone (organ-rem)	-	-	-	-	9×10^{-2}	-	-	-	-	-	9×10^{-2}	
<u>River Model "B"</u>												
Time = 1,000 years												
Thyroid (organ-rem)	-	-	5×10^{-4}	-	-	-	-	-	-	-	5×10^{-4}	
Time = 30,000 years												
Whole Body (person-rem)	-	-	-	-	-	-	6×10^{-1}	-	-	-	6×10^{-1}	
Time = 100,000 years												
Bone (organ-rem)	-	-	-	-	3×10^{-2}	-	-	-	-	-	3×10^{-2}	

^a All results are 50-year individual or population dose commitments resulting from one-year ingestion intake.

^{**} Population consists of 250 people served from a single water source.

^{***} Population consists of 10,000 people served from a single water source.

[†] Due to rounding of nuclide values to one significant figure after totaling, totals shown may not agree with nuclide values shown.

[‡] Maximum individual and upper limit population doses represent extreme conditions.

Note: Dashes are shown where nuclide dose values have a small or negligible contribution to the total dose commitment.

TABLE C-24

DOSE COMMITMENT CALCULATED FOR AQUEOUS TRANSPORT BY
FLOW INTO A POTABLE AQUIFER (SCENARIO 2)*

	Fission Products				Radium and Actinides						
	Sr-90	Tc-99 (R _d = 0)	I-129	Cs-137	Ra-226	U-234	Np-237	Pu-240	Pu-242	Am-243	TOTAL [†]
"Maximum" Individual Dose[†]											
Time = 300 years											
Whole Body (rem)	4x10 ²	3x10 ⁻¹	2x10 ⁻¹	3x10 ¹	2x10 ⁻¹	-	3x10 ⁰	9x10 ⁰	-	9x10 ⁻¹	5x10 ²
Bone (rem)	2x10 ⁴	8x10 ⁻¹	-	7x10 ¹	2x10 ⁻¹	-	8x10 ¹	4x10 ²	-	1x10 ¹	2x10 ⁴
Thyroid (rem)	-	-	1x10 ²	-	-	-	-	-	-	-	1x10 ²
Time = 6,000 years											
Whole Body (rem)	-	-	-	-	3x10 ²	1x10 ⁰	7x10 ⁻¹	4x10 ¹	4x10 ⁻¹	4x10 ⁰	3x10 ²
Bone (rem)	-	-	-	-	3x10 ²	2x10 ¹	2x10 ¹	3x10 ³	2x10 ¹	6x10 ¹	2x10 ³
Thyroid (rem)	-	-	6x10 ⁰	-	-	-	-	-	-	-	6x10 ⁰
"Maximum" Population Dose^{**} †											
Time = 300 years											
Whole Body (person-rem)	6x10 ⁴	4x10 ¹	2x10 ¹	4x10 ³	2x10 ¹	-	4x10 ²	1x10 ³	-	-	6x10 ⁴
Bone (organ-rem)	3x10 ⁶	1x10 ²	-	8x10 ³	2x10 ¹	-	1x10 ⁴	5x10 ⁴	-	-	3x10 ⁶
Thyroid (organ-rem)	-	-	2x10 ⁴	-	-	-	-	-	-	-	2x10 ⁴
Time = 6,000 years											
Whole Body (person-rem)	-	-	-	-	4x10 ⁴	1x10 ²	9x10 ¹	5x10 ³	5x10 ¹	5x10 ²	5x10 ⁴
Bone (organ-rem)	-	-	-	-	4x10 ⁴	2x10 ³	2x10 ³	2x10 ⁵	2x10 ³	7x10 ³	2x10 ⁵
Thyroid (organ-rem)	-	-	7x10 ²	-	-	-	-	-	-	-	7x10 ²
"Reasonable" Population Dose^{***}											
River Model "A"											
Time = 500 years											
Thyroid (organ-rem)	-	-	2x10 ⁴	-	-	-	-	-	-	-	2x10 ⁴
Time = 6,000 years											
Whole Body (person-rem)	-	-	-	-	-	-	1x10 ²	-	-	-	1x10 ²
Time = 100,000 years											
Bone (organ-rem)	-	-	-	-	1x10 ⁰	-	-	-	-	-	1x10 ⁰
River Model "B"											
Time = 500 years											
Thyroid (organ-rem)	-	-	1x10 ³	-	-	-	-	-	-	-	1x10 ³
Time = 6,000 years											
Whole Body (person-rem)	-	-	-	-	-	-	2x10 ¹	-	-	-	2x10 ¹
Time = 100,000 years											
Bone (organ-rem)	-	-	-	-	4x10 ⁻¹	-	-	-	-	-	4x10 ⁻¹

* All results are 50-year individual or population dose commitments resulting from one-year ingestion intake.

** Population consists of 250 people served from single water source.

*** Population consists of 10,000 people served from single water source.

† Due to rounding of nuclide values to one significant figure after totaling, totals shown may not agree with nuclide values shown.

† Maximum individual and upper limit population doses represent extreme conditions.

Note: Dashes are shown where nuclide dose values have a small or negligible contribution to the total dose commitment.

TABLE C-25

SUMMARY OF TOTAL
"MAXIMUM" INDIVIDUAL AND POPULATION
DOSE COMMITMENTS FOR DIRECT INGESTION OF CONTAMINATED AQUIFER

	Scenario 1			Scenario 2		
	Diffusion to Aquifer			Aqueous Transport to Aquifer		
	300 Years	30,000 Years		300 Years	6,000 Years	
I. "Maximum" Individual Dose Commitment ⁺						
A. Whole Body Dose (rem)	9×10^{-9}	6×10^1		5×10^2	3×10^2	
B. Bone Dose (rem)	4×10^{-7}	1×10^2		2×10^4	2×10^3	
C. Thyroid Dose (rem)	4×10^{-8}	$(6 \times 10^{-5})^*$		1×10^2	6×10^0	
II. "Maximum" Population** Dose Commitment ⁺						
A. Whole Body Dose (person-rem)	1×10^{-6}	8×10^3		6×10^4	5×10^4	
B. Bone Dose (organ-rem)	5×10^{-5}	2×10^4		3×10^6	2×10^5	
C. Thyroid Dose (organ-rem)	1×10^{-5}	$(7 \times 10^{-3})^*$		2×10^4	7×10^2	

* These are values for 1,000 years, which decrease with increasing time.

**Population consists of 250 people served from single source.

+ "Maximum" individual and population doses represent extreme conditions.

body. Strontium-90 would be the nuclide responsible for this high whole-body radiation dose and also for the calculated 20,000 rem to the bone. The contributions from the actinide elements at 300 years are appreciable but still small compared with the dose commitments that would be produced by the fission products.

After 6,000 years of direct flow through the repository, the actinide elements would reach maximum concentrations in the aquifer above the repository. At this time, the "maximum" individual dose commitment would be 300 rem to the whole body and 2×10^4 rem to bone. Under those conditions, the major contributors to the calculated doses are Ra-226 for the whole body and Ra-226 and Pu-240 for bone. The dose commitment to the thyroid at 300 years is 100 rem, which then decreases continuously with increasing time.

C-5.3.3 "Maximum" Population Radiation Dose

C-5.3.3.1 Direct Access to a Potable Aquifer

The underground water flow rates developed in Section C-4 are such that the maximum population that could be supplied by the contaminated water in the aquifer above the repository would be about 250 people.* The "maximum" population dose commitments were calculated, therefore, as the product of the "maximum" individual dose commitments multiplied by 250, except that the value used for the individual dose is reduced by a factor of two to allow for an assumed average individual water intake of only 370 ℓ /year. These are the values shown in Tables C-23, C-24 and C-25. A sample calculation is shown in Appendix C-IV. If the aquifer volume were larger, a larger population might be supplied, but the radionuclide concentrations would be decreased proportionately. Consequently, the "maximum" population dose commitment is bounded by the assumptions of initial transport of nuclides from the repository into the potable aquifer.

C-5.3.3.2 Diffusion to a Potable Aquifer (Scenario 1)

For the diffusion case, (Table C-23) calculated population dose commitments are extremely small at 300 years, i.e., 1×10^{-6} person-rem

*The assumption of 250 people supported by this aquifer is based on a water flow rate of 1.44×10^5 ℓ /day (Section C-4.4.3.2) and an individual water intake rate of 370 ℓ /year.

to the whole body; 5×10^{-5} organ-rem to bone; and 1×10^{-5} organ-rem to the thyroid. After 30,000 years, when the actinides have reached maximum concentrations, the whole-body population dose commitment would be 8×10^3 person-rem, mainly due to Ra-226, and the population dose commitment to bone would be 2×10^4 organ-rem, mainly due to Ra-226, Pu-239, and -240. The dose commitment to the thyroid at 300 years would be 1×10^{-5} organ-rem, rising to 7×10^{-3} organ-rem at 1000 years and decreasing thereafter to become negligible with respect to the other dose commitments.

C-5.3.3.3 Flow to a Potable Aquifer (Scenario 2)

For the flow case, the "maximum" population dose commitments are on the order of tens of thousands of person-rem or organ-rem per year of intake. The largest calculated value is 3×10^6 organ-rem to bone at 300 years, entirely due to strontium-90. The bone dose commitment at 6000 years was calculated to be 2×10^5 organ-rem, primarily from plutonium-240, americium-243, and some radium-226.

It is not the purpose of this task to go beyond dose calculations and project health effects. It should be emphasized, however, that any translation of "maximum" individual or population dose commitments to actual health effects based on the BEIR report's prediction of effects of low levels of radiation on large populations⁽⁵⁰⁾ are inappropriate. The maximum doses calculated for Scenario 2 are such that, if those conditions occurred, the health effects would be acute rather than long term. Furthermore, the size of the population would be too small to justify the use of large-population statistics. Instead of multiplying the total number of person-rem or organ-rem by the risk factors indicated in the BEIR report, it would be more realistic to conclude that, at these concentrations, intake by 250 people over a period of a few years would result in 250 fatalities.

Under the "worst case" conditions assumed in Scenario 2, "normal" adsorption was assumed in the aquifer, but none in the repository and the communicating channels to the aquifer. If "normal" adsorption were to be assumed in either or both of the latter two regions, then the

dose commitments to these 250 people would be reduced substantially, and any impact on them would probably be a long term, rather than acute, one.

C-5.3.4 "Reasonable" Population Dose

On the basis of the assumptions stated earlier, i.e., discharge of the contaminated aquifer into one of two river models at a distance of one mile down gradient from the repository, and for the postulated condition in which the only movement of radionuclides is by aqueous diffusion (Scenario 1), the calculated population dose commitments are in the range of 8×10^{-3} organ-rem to the thyroid after 1000 years to 3 person-rem to the total body after 30,000 years. These values are shown in the lower half of Table C-23 and in Table C-26. Sample calculations for the four pathways considered are included in Appendix C-V.

For the postulated flow case (Scenario 2), the calculated population dose commitments are approximately three orders of magnitude higher than for Scenario 1. (Note, however, that the differences between population dose commitments calculated for Scenarios 1 and 2 are not as great for the "reasonable" population dose commitments as they are for the "maximum" population dose commitments.) The population dose commitments to the thyroid are estimated to be 2×10^4 organ-rem after 500 years for River Model "A," and 1000 organ-rem after 500 years for River Model "B," as shown in the lower half of Table C-24 and in Table C-26. Although not shown in the tables, the population dose commitments from iodine-129 would remain constant over the times shown for other nuclides: 6000, 30,000, or 100,000 years. The whole-body population dose commitments for either of the river models reach a maximum at approximately 6000 years and are attributable to neptunium-237. The maximum population bone dose commitments are calculated to result from radium-226, which does not reach its maximum by ingrowth from uranium and thorium until at least 100,000 years.

The total population dose commitments from all pathways are consistently lower for River Model "B" than for River Model "A" (Tables C-23 and C-24). The contributions to population dose by individual

TABLE C-26

SUMMARY OF REASONABLE POPULATION DOSE COMMITMENTS
BY SIGNIFICANT RADIONUCLIDES AND PATHWAYS

	Scenario 1		Scenario 2	
	$^{129}\text{I}^*$ (1,000 y)	$^{237}\text{Np}^+$ (30,000 y)	$^{129}\text{I}^*$ (500 y)	$^{237}\text{Np}^+$ (6,000 y)
				$^{226}\text{Ra}^\dagger$ (100,000 y)
River Model "A"				
Direct Ingestion	3×10^{-4}	4×10^{-1}	6×10^2	1×10^1
Ingestion of Crops	8×10^{-3}	3×10^0	2×10^4	9×10^{-1}
Ingestion of Beef	8×10^{-6}	3×10^{-4}	2×10^1	1×10^{-2}
Ingestion of Milk	1×10^{-4}	6×10^{-5}	3×10^2	5×10^{-2}
Total: All Pathways	8×10^{-3}	3×10^0	2×10^4	1×10^0
River Model "B"				
Direct Ingestion	5×10^{-4}	6×10^{-1}	1×10^3	2×10^1
Ingestion of Crops	1×10^{-5}	4×10^{-3}	3×10^1	2×10^{-1}
Ingestion of Beef	2×10^{-8}	1×10^{-6}	4×10^{-2}	5×10^{-5}
Ingestion of Milk	7×10^{-6}	6×10^{-6}	2×10^1	2×10^{-4}
Total: All Pathways	5×10^{-4}	6×10^{-1}	1×10^3	2×10^1

*Thyroid dose commitment (organ-rem).

+Whole body dose commitment (person-rem).

†Bone dose commitment (organ-rem).

pathways, however, are in some cases larger for River Model "B" than for River Model "A" (Table C-26). For River Model "A," the population dose commitments are dominated by the ingestion of crops; for River Model "B," direct ingestion of water contributes essentially all of the population dose commitment. All of these "reasonable" calculated dose commitments are actually unrealistically high because they assume that drinking water is not pretreated and that vegetables are neither washed nor cooked. The differences between the relative dose contributions from direct ingestion of water and from ingestion of crops are also greatly influenced by the assumptions that are made concerning the use of the river water. In both cases, however, the more indirect pathways (meat and milk consumption) do not contribute significantly to the calculated dose commitment.

C-5.4 EVALUATION OF DOSE-TO-MAN CONSIDERATIONS

Besides the specific calculations for the various radiation dose categories presented in the preceding sections, other evaluations are made in the following sections to bring the overall scope of the radioactive waste problem into focus. For example, I-129, because of its very long half-life, should be considered beyond its initial entry into the biosphere. An ultimate limit to the population dose from I-129 would be achieved only if it were all diluted by the stable iodine present in some fraction of the oceans' volume.

Naturally-occurring radium provides some clues as to the anticipated behavior of actinides in the biosphere. Little radium reaches the biosphere and average radiation doses to humans from radium are extremely small in spite of the large amounts in soils and rocks.

Comparison of the impact of the "most likely" leakage from an HLW repository with that calculated for sources of natural radioactivity indicates a very small impact from the waste in comparison with that from natural sources.

C-5.4.1 Worldwide Impact of Iodine-129

Of the fission products contained in HLW, the one that seems most likely to reach the biosphere as a result of leaching by groundwater is I-129. This conclusion is based on the long half-life of I-129 (17 million years) and the relatively high mobility of iodine (compared with the few other long-lived fission products) in groundwater. The 50-year dose

commitment to the first generation exposed has been estimated by assuming typical transfer coefficients to a population located in the vicinity of the release point.

The question of ultimate population dose commitment is also of interest. This concept has been called the "environmental dose commitment,"⁽⁵⁵⁾ although the dose referred to is to human populations and not the "environment." The ultimate population dose commitment can be defined as the sum of the doses to all individuals at all future times that will result from a specified release of radioactivity to the environment. Although extremely easy to define conceptually, this dose commitment is impossible to calculate meaningfully. The only quantitative data available for the calculation are the physical decay rates for each radionuclide and the abundances and relative distributions of stable elements in the biosphere.

The uncertainties involved in making estimates of future population distributions, agricultural practices, etc., have been discussed in Section C-5.1.3. Even if one assumes that human populations and the general biosphere will be the same at all future times as they are now, calculations of ultimate dose commitments are limited by a lack of data on environmental dispersion rates. The population dose rate at any time depends only on the total amount of radioactivity in human beings at that time and not on the number of people actually exposed or in the total population. (This assumes that age-dependent dose factors are averaged over sufficiently large populations and are always in the same proportions.) Calculations of ultimate dose commitments, therefore, involve the estimation of the fraction of released radioactivity that will be available to, or actually in, human beings at any future time.

The ultimate limit on population doses from I-129 released to the biosphere would most likely be due to the ultimate dilution by stable iodine in the earth's crust and in the oceans. The mobility of iodine in water is indicated by the fact that the iodine is more easily leached and transported underground than other nuclides. The same relative mobility could be assumed in surface waters, indicating that most of the iodine would eventually flow into the oceans. The average concentration of stable iodine in the oceans is 0.06 ppm (g/m^3).⁽¹⁸⁾ The total area of the oceans is $361 \times 10^6 \text{ km}^2$ ⁽⁵⁶⁾ and, if one assumes a mixing depth of 0.1 km, the total volume available for mixing would be $36 \times 10^6 \text{ km}^3$. At a mean concentration of 0.06 g/m^3 , this top 0.1-km layer of the

oceans' surfaces would contain approximately 2×10^{15} g of stable iodine. This is approximately 10^8 times more than the total amount of I-129 projected to be contained in geologic repositories (Table C-9) as a result of nuclear power generation. Instead of the intrinsic specific activity of 1.8×10^8 pCi/g, the effective specific activity of I-129 would be reduced by roughly eight orders of magnitude to 2 pCi/g. With a normal annual iodine intake of about 0.07 g (200 µg/day), the total annual intake would be approximately 0.14 pCi.

The average individual thyroid dose from I-129 (if it returned to human beings after uniform dilution by the stable iodine in the ocean) would be 1×10^{-6} rem/year or about 5×10^{-5} rem in 50 years. For a world population of 4 billion, the population dose commitment would be about 4×10^3 organ-rem/year or 2×10^5 organ-rem in 50 years. The I-129 could be assumed to be diluted into a smaller volume of seawater (e.g., the top 0.1 km over only 1% of the earth's ocean surfaces). Under such an assumption, the average thyroid dose commitment to an individual could be larger, but the population exposed would be proportionately smaller, so the total population dose commitment calculated under such assumptions would not be greatly different from the previous calculation.

As pointed out in an Environmental Protection Agency (EPA) report on the concept of environmental dose commitment, no data are available from which to base projections of redistribution of elements in the biosphere over extended periods of time.⁽⁵⁵⁾ All calculations contained in this report are based on assumptions considered to be conservative with respect to the biosphere as it exists today, and the estimated dose commitments are believed to be upper limits for any reasonable assumptions of redistribution. All calculations are made for the most direct pathways to achieve maximum exposures and hence dose commitments for any time after release of material from the repository.

C-5.4.2 Comparison with Natural Background Radiation

If radioactive wastes from a geologic repository are contacted and leached by underground water, the subsequent transfer to the biosphere and to human beings may be assumed to be similar to that of naturally-occurring radionuclides. Radium is ubiquitous in the earth's crust and occurs with a wide range of concentrations in many surface streams.⁽⁵⁷⁾ Uranium ore of commercial grade typically contains radium at concentrations of hundreds of pCi/g, often in contact with groundwater. Sedimentary rock and many surface soils contain radium concentrations in the range of 0.1 to at least 1 pCi/g.

Radium is less mobile in the biosphere than many lighter elements, including most fission products, but is considerably more mobile than most of the actinide elements that comprise the major portion of the very long-lived materials in radioactive wastes. Environmental transfer data for the radionuclides of greatest significance are listed in Table C-20. The fact that the values for the actinides (represented by neptunium-237) are generally smaller than those for radium indicates the lesser mobility of these nuclides in environmental media. Furthermore, the fractional uptake of the actinides from the human gastrointestinal tract to blood is estimated by the International Commission on Radiological Protection to be 3×10^{-5} , compared with 0.3 for radium.⁽⁵⁸⁾ In this respect, the behavior of radium in the biosphere constitutes a very conservative basis for comparison of the projected radiation doses that would result from environmental contamination by actinides.

The extent to which radium in the environment is transferred to people is indicated by data contained in a recent report of the National Council on Radiation Protection and Measurements.⁽⁵⁷⁾ The mean dietary intake of radium from foods and drinking water is estimated to be 1.4 pCi/day. The mean skeletal content, based on analysis of bone ash, is approximately 5×10^{-3} pCi/g of bone. The same report provides an absorbed dose conversion factor of $200 \text{ mrad yr.}^{-1}/\text{pCi g}^{-1}$ in bone.⁽⁵⁷⁾ If a conservative value for the quality factor is assumed to be 10 for radium and its decay products, the dose-equivalent conversion factor becomes $2 \text{ rem yr.}^{-1}/\text{pCi g}^{-1}$ for bone. Thus the mean skeletal dose equivalent to individuals in the United States is estimated to be: $5 \times 10^{-3} \text{ pCi/g} \times 2 \text{ rem yr.}^{-1}/\text{pCi g}^{-1}$, or approximately 10^{-2} rem/year. For approximately 200 million persons, the population bone dose equivalent is approximately 2×10^6 organ-rem/yr. or 10^8 organ-rem for 50 years.

For the postulated transport of waste materials from a repository by flow (Scenario 2), the maximum population 50-year dose commitment to bone was calculated to be 3×10^6 organ-rem for an intake of one year (Table C-24). For a 50-year intake, the bone dose commitment amounts to 1.5×10^8 organ-rem, a value comparable to that from radium in the natural environment. This calculation, of course, represents a "worst case" in that it involves the direct consumption of water that has flowed directly through the waste repository into the aquifer and has subsequently undergone a minimum loss of radionuclides by dilution or adsorption.

The total radiation dose to bone from natural sources of radiation includes external sources (cosmic and terrestrial) in addition to the internal deposition of radium and other radionuclides. The total radiation bone dose from natural sources is comparable to the whole-body radiation dose and is on the order of 0.1 rem/yr. to average individuals in the United States population.⁽⁵⁷⁾ The total population dose commitment to bone for 200 million people is, then, approximately 2×10^7 organ-rem/yr. and 10^9 organ-rem/50 years.

As indicated in previous sections of this report, the quantities of actinides that could be expected to reach the biosphere under any reasonable scenario for leaching by groundwater, would be orders of magnitude smaller than the activity of radium in the biosphere, even in the immediate vicinity of a geologic waste repository. Although it is possible to hypothesize extreme scenarios for which actinides (as well as fission products) could reach the biosphere in relatively high concentrations, the number of people who might be exposed to such concentrations would be small. The largest population dose commitment to the whole body, calculated for Scenario 2 at 300 years, is 6×10^4 person-rem for a 1-year intake. This number is equivalent to 0.3% of the annual population dose commitment from natural radiation sources. For all other expected times or modes of release of radionuclides from a repository, (i.e., transport by flow at other times or diffusion at any time) the upper-limit population dose commitments are smaller. Hence, the 50-year dose commitments calculated for "reasonable scenarios," i.e., by contaminated surface water and irrigated food pathways, are negligible in comparison with radiation doses from natural sources.

C-5.4.3 Comparison With Radium-226 Currently Present in the Colorado River

A useful perspective is obtained by comparing the Ra-226 population dose commitment calculated for Ra-226 concentrations presently occurring in the Colorado River with that obtained for Ra-226 using Scenario 2 (aqueous flow) and River Model "A". To make this comparison, actual Ra-226 concentrations in the Colorado River were used as input parameters to River Model "A". Dose commitments for Ra-226 for Scenario 2 using River Model "A" were calculated as presented in Section 5.2.3.2.

Table C-27 indicates the results of calculations that compare the Colorado River with Scenario 2, River Model "A". The total dose commitment from all pathways from the Colorado River is 5×10^4 organ-rem, while the dose commitment from Scenario 2, River Model "A" is 1 organ-rem. According to this calculation, the Colorado River now contributes 10,000 times more radiation dose to the present population than would an HLW repository at the time of maximum concentration of Ra-226 in the biosphere to a population existing in the same location 100,000 years following waste emplacement.

The dose commitments calculated for the Colorado River are conservative because they do not account for radium removal during water treatment to remove hardness, which typically occurs before municipal utilization of this water. Nevertheless, the projected radiation doses from the proposed HLW release are still quite small compared with those arising from the Colorado River now.

TABLE C-27

COMPARISON OF "REASONABLE" POPULATION
50-YEAR DOSE COMMITMENTS FOR Ra-226 FROM
SCENARIO 2 AND Ra-226 IN THE COLORADO RIVER

Exposure Pathway	Ra-226 Dose Commitment*	
	Colorado River (organ rem)	Scenario 2 River Model "A"+ (organ rem)
Direct Ingestion	2×10^4	3×10^{-1}
Ingestion of Crops	3×10^4	9×10^{-1}
Ingestion of Beef	2×10^2	3×10^{-2}
Ingestion of Milk	5×10^2	5×10^{-2}
Total: All Pathways	5×10^4	1×10^0

*Radium dose commitments for the Colorado River are based on actual radium concentrations in the Colorado River at the present time used as input to River Model "A".

+Results for River Model "A" are the same as presented in Table C-26 for Scenario 2; the time period is 100,000 years following waste emplacement.

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APPENDIX C-I

GLOSSARY

Actinides: A series of elements in the periodic table, beginning with actinium (element No. 89) and continuing through lawrencium (element No. 103). The series includes uranium (element No. 92) and all of the man-made transuranium elements. All are radioactive.

Aquifer: Rock or soil strata containing water that reaches the ground surface in any natural or man-made manner.

Argillaceous: Referring specifically to argillite, a rock formed from clays, mud, and silt, but more compacted and solidified than shale or mudstone. Although the distinction between shale and argillite is imprecise, it is used here to imply that the rock has sufficient strength and rigidity to be mined, and that it has not been metamorphosed into slate, which has a platy cleavage.

Background radiation: The radiation in man's natural environment, including cosmic rays and radiation from the naturally radioactive elements, both outside and inside the bodies of men and animals. It is also called natural radiation.

Basalt: A hard, dense, dark fine-grained igneous rock.

Bedded salt: Salt occurring in strata, commonly interspersed with layers of shale or limestone.

Biosphere: Zone at and adjacent to the earth's surface where all life exists.

Breccia pipe: A geologic formation composed of broken rock, breccia, filling a vertical cylindrical hole, or pipe. Such structures have also been identified as Collapse Structures, Pseudo-Karst Formations and Dry Sinkholes. Their development is assumed to require the dissolution of underlying rock by circulating groundwater, and collapse of the rock above into the cavity thus formed.

Curie: The basic unit to describe the intensity of radioactivity in a material. The curie is equal to 37 billion disintegrations per second, which is approximately the rate of decay of 1 gram of radium. A curie is also a quantity of any nuclide having 1 curie of radioactivity. The prefixes milli-, micro-, nano-, and pico- are frequently used and indicate quantities of 10^{-3} curie, 10^{-6} curie, 10^{-9} curie, and 10^{-12} curie, respectively.

Darcy's Law: The relationship between the hydraulic gradient and the groundwater velocity.

Diapirism: The forceful intrusion of one geologic material into another, overlying material.

Distribution coefficients (K_d) and (K_a): Sorptions or mechanisms causing ions to move at lower velocities than the groundwater transporting them. K_d = distribution coefficient for porous media; K_a = distribution coefficient for faulted or fractured media.

Fault: A break in the continuity of a rock formation, caused by a shifting or dislodging of the earth's crust, in which adjacent surfaces are differentially displaced parallel to the plane of fracture.

Geosphere: The solid earth, including rocks, soil, and water in the ground, but excluding oceans (hydrosphere) and air (atmosphere).

Half-life (radioactive): The time required for one-half of an initially radioactive material to undergo nuclear transformation; the half-life is a measure of the persistence of radioactivity and is unique to each radionuclide.

High-level waste: The highly radioactive waste resulting from the reprocessing of spent fuel to separate uranium and plutonium from the fission products. The term includes the high-level liquid wastes (HLLW) produced directly in reprocessing, and the solid high-level wastes (HLW) which can be made therefrom.

Hydraulic gradient: The pressure drop per unit length of distance traveled.

Igneous rocks: A group of rocks formed by the solidification of molten rock. The group consists of intrusive (plutonic) rocks which have solidified below the earth's surface and extrusive (volcanic) rocks which solidified after eruption.

Interstitial velocity: Groundwater flow through the pores and openings and around individual grains of geologic media.

Ion: An atom or molecule that has lost or gained one or more electrons. By this ionization, it becomes electrically charged. Examples: an alpha particle, which is a helium atom minus two electrons; a proton, which is a hydrogen atom minus its electron.

Isotope: One of two or more atoms with the same atomic number (the same chemical element) but with different atomic weights. Isotopes have very nearly the same chemical properties, but different nuclear (radioactive-decay) properties. Thus, for the element carbon, for example, the isotope of atomic weight 12 (C-12) and the isotope of atomic weight 14 (C-14) behave identically in chemical reactions; but whereas C-12 is not radioactive, C-14 is radioactive, decaying with a 5730-year half-life to stable nitrogen (N-14) with release of a beta particle.

Laminar flow: Non-turbulent flow of fluid in layers near a boundary.

Limestone: A carbonate rock composed primarily of calcite, usually with varying amounts of dolomite.

Lithology: The study, description and classification of rock.

Nuclide: A general term applicable to all atomic forms of the elements. The term is often used erroneously as a synonym for "isotope," which properly has a more limited definition. Whereas isotopes are the various forms of a single element (hence are a family of nuclides) and all have the same atomic number and number of protons, nuclides comprise all the isotopic forms of all the elements.

Oxidation-reduction potential: Energy gained in the transfer of one mole of electrons from an oxidant to hydrogen, expressed in volts.

Permeability: The ability of a fluid to move through a medium under a hydraulic gradient.

Radioactivity: The spontaneous decay or disintegration of an unstable atomic nucleus, usually accompanied by the emission of ionizing radiation.

Radioisotope: A radioactive isotope. An unstable isotope of an element that decays or disintegrates spontaneously, emitting radiation. More than 1300 natural and artificial radioisotopes have been identified.

Radionuclide: A radioactive nuclide. Thus, carbon-14 (C-14) is a radionuclide because it decays radioactively to nitrogen-14 (N-14).

Radwaste: A contraction of the term "radioactive waste."

Redox potential: Oxidation-reduction potential (q.v.).

Rem: A dose unit which takes into account the relative biological effectiveness (RBE) of the radiation. The rem ("roentgen equivalent man") is defined as the dose of a particular type of radiation required to produce the same biological effect as one roentgen of (0.25 Mev) gamma radiation. A 1-rad dose of alpha particles is approximately equivalent in its biological effects to 10 rads of gamma radiation, and hence may be expressed as 10 rems. A millirem (mrem) is one thousandth of a rem.

Retardation factor (R_d): Ratio of the water velocity to the nuclide migration velocity.

Salt dome: A geologic salt formation in which a plug of salt has been thrust up through rock at some depth, leading to a subterranean "cylinder" of salt which may be a mile or more in diameter and several miles deep.

Shale: Lithified clay or mud, usually containing some sand and silt.

Seismicity: Caused by an earthquake or earth vibration.

Stratigraphy: The study of rock strata, especially of their distribution, deposition and age.

Styolite: A small columnar rock development in limestone and other calcareous rocks.

Tectonic: Pertaining to the formation of the earth's crust; the forces involved in or producing such deformation and the resulting forms.

Tuff: A rock composed of compacted volcanic ash varying in size from fine sand to coarse gravel.

APPENDIX C-II

RADIOACTIVE WASTE MIGRATION MODEL

The radioactive waste migration model used in this study is a single-phase, numerical model for transport of fluid, energy, salinity, and radioactive nuclides in porous media⁽²⁹⁾. Basic laws of physics are used to describe mathematically the nuclide transport in geologic formations. The model includes radioactive decay and generation (by decay of other components) and adsorption of nuclides.

The complete "flow system" is described by three coupled partial differential equations for conservation of total fluid mass, energy, and an inert component (e.g. brine). The equations are coupled through fluid properties--density and viscosity. Calculation of radioactive waste transport, in addition to the above three equations, requires separate solutions for each of the radioactive constituents. Conservation of mass of the species dissolved in the fluid phase and adsorbed on the rock as well as radioactive decay and generation from other nuclides are included.

Basic assumptions and model features can be summarized as follows:

- (1) Three-dimensional, transient, laminar flow.
- (2) Three-dimensional Cartesian (x, y, z) or two-dimensional (r, z) coordinate description.
- (3) Fluid density can be a function of pressure, temperature and concentration of the inert component. Fluid viscosity can be a function of temperature and concentration.
- (4) Waste dissolved in fluid is completely miscible with the in-place fluids.
- (5) Aquifer properties are varied with position.
- (6) Hydrodynamic dispersion is described as a function of fluid velocity.
- (7) The energy equation can be described as "enthalpy in - enthalpy out = change in internal energy of the system". This is rigorous, except for kinetic and potential energy, which have been neglected.
- (8) Boundary conditions allow natural water movement in the aquifer, heat losses to the adjacent formations,

and the location of discharge, withdrawal, and observation points within the system. Pressures, temperatures, and concentrations at these points are related to corresponding grid block centers through a wellbore model.

- (9) Fluid properties are independent of radioactive nuclide concentrations (trace quantities).
- (10) Radioactive nuclide source rates can be described through a leach model integrated in the waste migration model. The leach rate can be described by a logarithmic or a linear equation. The radionuclide concentration in fluid can be limited by solubilities.
- (11) Equilibrium adsorption and desorption exist. The equilibrium adsorption constant is related to desorption distribution constant, rock density, and porosity. The adsorption distribution constant is a function of rock or formation type only.

Suppose x, y, z to be a Cartesian coordinate system and let $Z(x,y,z)$ be the height of a point above a horizontal reference plane. Then the basic equation describing single-phase flow in a porous medium results from a combination of the continuity equation

$$\nabla \cdot \rho \mathbf{u} + q' = - \frac{\partial}{\partial t} (\epsilon \rho) \quad (I-1)$$

Net

Convection Source Accumulation.

and Darcy's law in three dimensions.

$$\mathbf{u} = - \frac{k}{\mu} (\nabla p - \rho g \nabla Z) \quad (I-2)$$

The result is the basic flow equation,

$$\nabla \cdot \frac{\rho k}{\mu} (\nabla p - \rho g \nabla Z) - q' = \frac{\partial}{\partial t} (\epsilon \rho) \quad (I-3)$$

The energy balance (defined as enthalpy in - enthalpy out = change in internal energy) is described by the energy equation,

* Detailed definitions of all terms are given in the Nomenclature.

$$\begin{aligned}
& \nabla \cdot \left(\frac{\rho k}{\mu} H (\nabla p - \rho g \nabla Z) \right) + \nabla \cdot \underline{R}_H \cdot \nabla T - q_L - q'H \\
& \quad \text{Net Energy} \quad \text{Conduction} \quad \text{Heat Loss} \quad \text{Enthalpy In} \\
& \quad \text{Convection} \quad \quad \quad \text{to Sur-} \quad \text{with Fluid} \\
& \quad \quad \quad \quad \quad \quad \text{rounding Strata} \quad \text{Source } q' \\
& - qH = \frac{\partial}{\partial t} [\epsilon \rho U + (1-\epsilon) (\rho C_p)_R T] \\
& \quad \text{Energy In} \quad \quad \quad \text{Accumulation} \\
& \quad \text{Without Fluid} \\
& \quad \text{Input}
\end{aligned} \tag{I-4}$$

A material balance for the solute results in the solute or concentration equation

$$\begin{aligned}
& \nabla \cdot (\rho \hat{C} \frac{k}{\mu} (\nabla p - \rho g \nabla Z)) + \nabla \cdot \rho \underline{E}_c \cdot \nabla \hat{C} - q' \hat{C} = \frac{\partial}{\partial t} (\rho \epsilon \hat{C}) \\
& \quad \text{Net Solute} \quad \quad \quad \text{Diffusion} \quad \quad \quad \text{Source} \quad \quad \quad \text{Accumulation} \\
& \quad \text{Convection}
\end{aligned} \tag{I-5}$$

A similar material for N radioactive components results in N component equations

$$\begin{aligned}
& \nabla \cdot [\rho C_{1\mu}^k (\nabla p - \rho g \nabla Z)] + \nabla \cdot \rho \underline{E}_c \cdot \nabla C_1 - q C_1 + \sum_{j=1}^N \kappa_{1j} R_j C_j \rho \epsilon \\
& \quad \text{Net Component 1} \quad \quad \quad \text{Diffusion of} \quad \quad \quad \text{Source of} \quad \quad \quad \text{Generation of} \\
& \quad \text{Convection} \quad \quad \quad \text{Component 1} \quad \quad \quad \text{Component 1} \quad \quad \quad \text{Component 1 by} \\
& \quad \quad \quad \quad \quad \quad \quad \quad \quad \quad \quad \quad \quad \quad \quad \text{decay of other} \\
& \quad \quad \quad \quad \quad \quad \quad \quad \quad \quad \quad \quad \quad \quad \quad \text{isotopes} \\
& - \sum_{k=1}^N \kappa_{k1} R_k \rho \epsilon C_1 = \frac{\partial}{\partial t} (\epsilon \rho R_1 C_1) \\
& \quad \text{Net Decay of} \quad \quad \quad \text{Accumulation} \\
& \quad \text{Component 1 to} \\
& \quad \text{Other Isotopes}
\end{aligned} \tag{I-6}$$

where

$$\kappa_{k1} R_1 \rho \epsilon C_1 = \kappa_{k1} \rho \epsilon C_1 + \kappa_{k1} \rho_s (1-\epsilon) C_s$$

and assumes the approximation

$$\frac{\partial}{\partial t} (\epsilon \rho R_1 C_1) = \frac{\partial}{\partial t} (\epsilon \rho C_1) + \frac{\partial}{\partial t} (1-\epsilon) \rho_s C_s$$

The equilibrium adsorption constant $R_1 = 1 + \frac{(1-\epsilon) \rho_s K_{d1}}{\epsilon}$

The system of equations, along with the fluid property dependence on pressure, temperature, and concentration, describe the reservoir flow due to discharge of wastes into an aquifer. This is a nonlinear system of partial differential equations, which must be solved numerically using high-speed digital computers. The equations are coupled with each other through fluid property dependence. Since the radioactive components are assumed to be present in trace quantities only, and the fluid properties are independent of these concentrations, the equations predicting the concentrations of radioactive components are uncoupled from the other equations.

The solution procedure is simple. First, the three differential equations for the flow system are solved over a numerical time step. Since the radionuclide transport equations are uncoupled from the first three equations, the calculated flow solution (pressures and velocities) is then used to solve the radioactive constituent equations. The radioactive nuclide transport equations are actually coupled through decay and generation terms. These are uncoupled by solving the equations in a systematic order, starting from parent constituent with no generation terms.

The model has been tested by comparing results with an analytical solution for a one-dimensional, homogeneous, constant-velocity, and three-component system⁽¹¹⁾. The solution for an individual decaying nuclide would have the form:

$$C = (C_0/2)e^{-kt} \left[\operatorname{erfc}\left(\frac{x-ut/\epsilon}{2\sqrt{E_{xx}}}\right) - e^{-\frac{ux}{E}} \operatorname{erfc}\left(\frac{x+ut/\epsilon}{2\sqrt{E_{xx}}}\right) \right] \quad (I-7)$$

For a step change, input superposition can be used with the above equation. A comparison of the numerical finite difference model with the analytical ORIGEN computer model for uranium-238 is shown in Figure C-II-1. The solid lines are ORIGEN results and the data points indicate the numerical model results. Table C-II-1 lists the nuclides, their half-lives and initial amounts present. Considering the range in component amounts and the time frame, the match between the two model results is excellent. This comparison indicates that the decay chains, including components with large contrast in half-lives, can be accurately described by the numerical model. Although this comparison deals only with decay, the same problem was solved for each nuclide migrative with the same degree of absorptivity. The comparison showed that the numerical model was capable of the same accuracy when dealing with migrations as with decay.

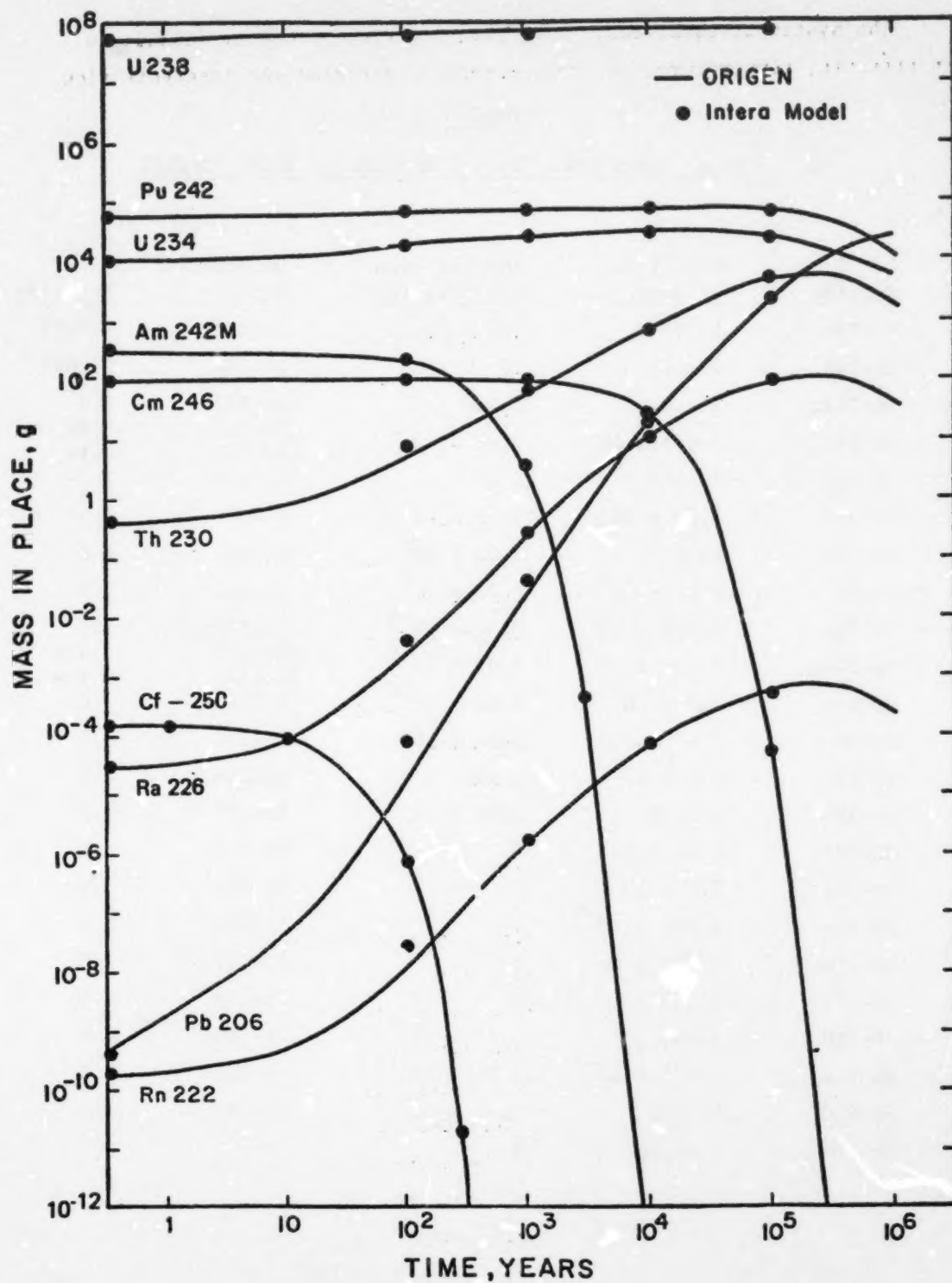


FIGURE C- II-1 COMPARISON OF ORIGEN AND INTERA MODEL RESULTS FOR STATIC FLOW CONDITIONS

TABLE C-II-1

DECAY CHAIN USED FOR COMPARISONS OF MODEL RESULTS

<u>Nuclide</u>	<u>Half-Life (Years)</u>	<u>Initial Mass Present, (g)</u>	<u>Daughter Nuclide</u>	<u>Fraction</u>
Cf-250	13.08	1.5×10^{-4}	Cm-246	0.9921
Cm-246	4711.0	101.0	Pu-242	0.9997
Am-242m	152.0	319.0	Am-242	1.0
Am-242	1.825×10^{-3}	3.83×10^{-3}	{ Cm-242	{ 0.82
Cm-242	0.4463	0.784	{ Pu-242	{ 0.18
Pu-242	3.79×10^5	6.15×10^4	Pu-238	1.0
Pu-238	89.0	1.26×10^4	U-238	1.0
U-238	4.51×10^9	5.73×10^7	U-234	1.0
Th-234	6.598×10^{-2}	8.32×10^{-4}	Th-234	1.0
Pa-234m	2.225×10^{-6}	1.93×10^{-8}	Pa-234m	1.0
Pa-234	7.7×10^{-4}	6.69×10^{-6}	{ Pa-234	{ 0.001
U-234	2.47×10^5	1.02×10^4	{ U-234	{ 0.999
Th-230	8.0×10^4	0.405	U-234	1.0
Ra-226	1600.0	2.94×10^5	Th-230	1.0
Rn-222	1.04×10^{-2}	0	Ra-226	1.0
Po-218	5.8×10^{-6}	0	Rn-222	1.0
Pb-214	5.095×10^{-5}	0	Po-218	1.0
Bi-214	3.746×10^{-5}	0	Pb-214	1.0
Po-214	6.338×10^{-12}	0	Bi-214	1.0
Pb-210	21.0	6.49×10^{-8}	Po-214	1.0
Bi-210	1.372×10^{-2}	2.94×10^{-11}	Pb-210	1.0
Po-210	0.3778	7.39×10^{-10}	Bi-210	1.0
Pb-206	Stable	0	Po-210	1.0
			Pb-206	1.0
			---	---

APPENDIX II

NOMENCLATURE

C	-	concentration of the radioactive/trace component
$\hat{C}$	-	concentration of the inert contaminant
C_s	-	concentration of the radioactive/trace component in adsorbed state
$\underline{E}$	-	dispersivity tensor (hydrodynamic + molecular)
g	-	acceleration due to gravity
H	-	fluid enthalpy
k	-	permeability
κ	-	constant decay
R_d	-	retardation factor
K_d	-	adsorption distribution constant
l	-	distance between adjacent grid block centers
p	-	pressure
q'	-	fluid source rate (withdrawal)
q_l	-	rate of heat loss
q_h	-	rate of energy withdrawal
t	-	time
T	-	temperature
T	-	transmissibility
$\underline{u}$	-	velocity vector
U	-	internal energy
Z	-	height above a reference plane
ϵ	-	porosity
μ	-	viscosity
ρ	-	fluid density
ρ_s	-	formation density
ρ_B	-	bulk density of rock and fluid

Subscripts

c - component (mass)
H - heat (energy)
R - rock (formation)
w - water (fluid)

APPENDIX C-III

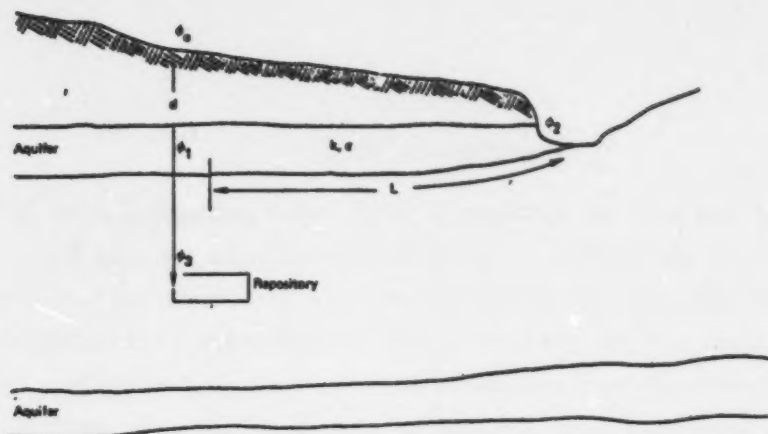
PARAMETERS OF STUDY

A meeting was held on January 9, 1979, with personnel from EPA, the Geological Survey and INTERA. The objective of this meeting was to establish reasonable bounds on the parameters (for such variables as permeability, porosity, flow gradients, and distribution coefficients) that could significantly affect geosphere transport of radionuclides. By this date, EPA had developed a probabilistic risk model for estimating the consequences associated with a high-level radioactive waste repository. In utilizing this risk model, sensitivity studies were to be performed covering a range of the important parameters. This appendix summarizes the results of these studies for comparison with the values listed in Tables C-7 and C-10.

The panel concluded that use of site criteria would establish at least three factors. These are:

- (1) Yields of ground water in aquifers or flow units adjacent to the repository site would be extremely low.
- (2) The original ground water environment in the vicinity of the repository would be a reducing one rather than an oxidizing one.
- (3) No organic material would be considered for storage in the high-level repository.

The simplified diagram below characterizes and defines the variables of interest in the geosphere pathways.



In the above diagram, ϕ_0 , ϕ_1 , ϕ_2 , ϕ_3 are the potentials at the surface, aquifer entry from repository, aquifer discharge, and repository, respectively. k_1 and ϵ_1 are the permeability and porosity of the aquifer at the depth from surface to repository, and L the pathway length in the aquifer.

The following summarizes the values for the above variables:

$\frac{L}{\text{m}}$	$\frac{k_1}{\text{cm/sec}}$	$\frac{\phi_1 - \phi_2}{L}$
$\geq 8 \text{ km (5 mi)}$	$10^{-5} - 10^{-3} \text{ cm/sec}$ ($2.8 \times 10^{-2} - 2.8 \text{ ft/day}$)	$10^{-4} - 10^{-1} \text{ m/m}$ (Note: 10^{-2} had almost equal support for the upper limit.)
$\frac{\epsilon_1}{\text{m}}$	$\frac{k_v}{\text{cm/sec}}$	$\frac{(\phi_0 - \phi_3)}{d}$
$0.001 - 0.3$	$\leq k_1 \text{ (Max)}$	$0 - 10^{-2} \text{ m/m}$

vertical permeability
from repository to surface

$\frac{\epsilon}{\text{m}}$
$0.01 - 0.3$
(porosity for vertical pathway)

In addition, the panel established the following limitations:

- (1) Travel time in the aquifer should be greater than 1000 years (siting criteria would eliminate lower travel times).
- (2) Adsorption coefficients should be assumed as follows:

<u>Nuclides</u>	R_d (Retardation Factor)
C-14	1
Kr	1
Tc	1
I-129	1
Sr	10, 100
Cs	10, 100
All actinides over a range of 1, 10, 1000 (fully correlated)	

- (3) Limit the nuclides leaving the repository by solubility in a reducing environment.

The above criteria (3) would undoubtedly limit by solubility Tc, and the actinide nuclides. It might also limit others depending upon anions present.

The above parameter ranges were provided by the panel to give guidance in the range of variables for EPA's sensitivity analysis. In some cases, e.g., the adsorptivities, other values have been measured and reported. However, investigation over the above range should provide guidance in establishing the range of nuclide concentrations that might reach the biosphere.

APPENDIX C-IV

"MAXIMUM" INDIVIDUAL AND POPULATION DOSE COMMITMENT SAMPLE CALCULATIONS

"MAXIMUM" INDIVIDUAL DOSE COMMITMENT

The "maximum" 50-year individual ingestion dose commitment is calculated as the product of the drinking water activity (pCi/l) times the maximum annual individual intake (liters) times the 50-year adult ingestion dose factor (rem/pCi). For ingestion of strontium-90 at 300 years in Scenario 2 (aqueous transport), the whole-body dose commitment is:

$$3.6 \times 10^6 \text{ pCi/l} \times 730\text{l} \times 1.66 \times 10^{-7} \text{ rem/pCi} = 4.4 \times 10^2 \text{ rem}$$

"MAXIMUM" POPULATION DOSE COMMITMENT

The "maximum" population 50-year ingestion dose commitment is calculated as the product of the drinking water activity (pCi/l) times the average annual individual intake (liters) times the population (persons) times the 50-year adult ingestion dose factor. For ingestion of strontium-90 at 300 years in Scenario 2 (aqueous transport), the whole-body dose commitment is:

$$\begin{aligned} 3.6 \times 10^6 \text{ pCi/l} \times 370\text{l} \times 250 \text{ persons} \times 1.66 \times 10^{-7} \text{ rem/pCi} \\ = 5.6 \times 10^4 \text{ person rem} \end{aligned}$$

APPENDIX C-V

"REASONABLE" POPULATION DOSE METHODOLOGY AND SAMPLE CALCULATION

Case: Aqueous Flow Transport (Scenario 2), River Model "A" for neptunium-237 at 6,000 years.

From Table C-19, Np-237 has a concentration in the aquifer at one mile downgradient of the repository equal to 7.4×10^{-5} g/l. Using the specific activity of Np-237 (7.3×10^{-4} Ci/g), the activity in the aquifer as it intersects the river is:

$$7.4 \times 10^{-5} \text{ g/l} \times 7.3 \times 10^{-4} \text{ Ci/g} = 5.4 \times 10^{-8} \text{ Ci/l or} \\ 54,000 \text{ pCi/l} \quad (1)$$

From Section C-4.4.2, the total discharge of the aquifer is 144,000 liters per day (0.06 cfs). The river discharge at the aquifer intersect (from the river model) is 2,400 ft³/sec (cfs). The initial activity of Np-237 in the river is then:

$$(1 \text{ l/day} = 4.1 \times 10^{-7} \text{ cfs}) \\ 54,000 \text{ pCi/l} \times \frac{0.06 \text{ cfs}}{2,400 \text{ cfs} + 0.06 \text{ cfs}} = 1.3 \text{ pCi/l} \quad (2)$$

A. Direct Ingestion Dose Commitment

The 50-year population dose commitment resulting from one-year ingestion of the river water is calculated as the product of the activity at the point of diversion for drinking water use (pCi/l) times the average individual's annual intake (l) times the population drinking the water (persons) times the 50-year adult ingestion dose factor for Np-237 (rem/pCi).

For the first population (i.e., that at the first diversion only), drinking water below the aquifer intersect, the values used are:

- Dilution factor = 1.7 (from river model)
- Activity of diversion = $1.3 \text{ pCi/l} \div 1.7 = 7.6 \times 10^{-1} \text{ pCi/l}$
- Average individual's annual intake = 370 liters (54)
- The population drinking river water = 1×10^4 persons (from the river model)
- The 50-year adult ingestion dose factor for Np-237 = $5.54 \times 10^{-8} \text{ rem/pCi}$ (54)

• The 50-year population dose commitment is:

$$7.6 \times 10^{-1} \text{ pCi/l} \times 370\text{l} \times 1 \times 10^4 \text{ persons} \times 5.54 \times 10^{-8} \text{ rem/pCi} = 0.2 \text{ person-rem} \quad (3)$$

The total population dose commitment resulting from direct ingestion of contaminated water is shown in Table C-26 for important radionuclides at the time of their maximum concentration in the model river. These dose commitments result from a summation of population dose commitments at each river diversion for each nuclide.

B. Ingestion of Contaminated Crops

The 50-year population dose commitment resulting from ingestion of one-year's crop irrigated with river water was calculated as the product of the crop activity (pCi/kg) times the annual garden vegetable yield (kg) times the ingestion dose factor. The crop activity for irrigated crops is the sum of the activity resulting from direct deposition on the vegetation and the activity resulting from uptake through the soil. The equation for estimating plant activity from irrigation follows: (54)

$$C_v = C_w I \left[\frac{r \left(1 - e^{-\lambda_E t} \right)}{Y_v \lambda_E} + \frac{B_v \left(1 - e^{-\lambda t_b} \right)}{P \lambda} \right] e^{-\lambda t_h} \quad (4)$$

where C_v = crop activity (pCi/kg)

C_w = irrigation water activity (pCi/l)

I = irrigation water application rate (l/m²-hr)

r = fraction of activity retained on crops (0.25 for sprinkler irrigation)

Y_v = agricultural yield (2.0 kg/m² for vegetation, 0.75 kg/m² for crop)

B_v = the transfer fraction from soil to vegetation for each element

$$\left(\frac{\text{pCi/kg in plant}}{\text{pCi/kg in soil}} \right)$$

P = the effective soil "surface density" (240 kg/m² for 15.24 cm (6-inch depth)

λ = radioactive decay constant (hr⁻¹) for each nuclide

λ_E = effective removal rate constant ($\lambda_E = \lambda + \lambda_w$ where λ_w is the weathering loss (0.0021 hr⁻¹)

t_e = time that crops are exposed to contamination during the growing season (hrs)

t_b = midpoint of soil exposure time (hrs)

t_h = time between harvest and consumption (hrs).

The first fraction in the brackets accounts for the plant activity resulting from direct deposition and the second fraction represents the soil uptake activity.

For all cases, the nuclides leaving the aquifer and contaminating the river have sufficiently long half-lives that their removal by radioactive decay in crops or soils is negligible. Consequently, the above equation may be simplified based on the following relationship: As a nuclide's half-life approaches infinity, λ approaches 0.

$$\lim_{\lambda \rightarrow 0} \lambda_E = \lambda_w = 0.0021 \text{ hr}^{-1} \quad (5)$$

$$\lim_{\lambda \rightarrow 0} \left[\frac{1 - e^{-\lambda t_b}}{\lambda} \right] = t_b \quad (6)$$

$$\lim_{\lambda \rightarrow 0} e^{-\lambda t_h} = 1 \quad (7)$$

The equation then reduces to:

$$C_v = C_w I \left[\frac{r \left(\frac{1 - e^{-\lambda_w t_e}}{\lambda_w Y_v} \right) + \frac{B_v t_b}{P}} \right] \quad (8)$$

The above equation must be expanded as follows to consider the 10% sprinkler and the 90% ditch irrigation contribution to the vegetation activity:

$$C_v = 0.1 \left\{ C_w I \left[\frac{r(1 - e^{-\lambda_w t_e})}{\lambda_w Y_v} + \frac{B_v t_b}{P} \right] \right\} + 0.9 \left\{ C_w I \left[\frac{B_v t_b}{P} \right] \right\} \quad (9)$$

The first term in the brackets accounts for the activity resulting from the deposition on plants and from soil uptake (sprinkler irrigation) and the second term in the brackets accounts for uptake from the soil only (ditch irrigation).

The following terms are constants in the above equation:

$$r = 0.25 \text{ (sprinkler irrigation)}$$

$$\lambda_w = 0.0021 \text{ hr}^{-1}$$

$$t_e = 1000 \text{ hrs}$$

$$P = 240 \text{ kg/m}^2$$

$$t_b = 50 \text{ years or } 4.4 \times 10^5 \text{ hrs}$$

For neptunium-237 at the diversion in River Model "A" corresponding to a 1.7 dilution factor, $C_w = 0.76 \text{ pCi/l}$, I (from river model) $= 0.23 \text{ l/m}^2\text{-hr}$, $B_v = 2.5 \times 10^{-3}$ (1), $Y_v = 2.0 \text{ kg/m}^2$ (1)

Evaluation of terms in equation (9) results in the following:

$$\frac{r(1-e^{-\lambda_w t_e})}{\lambda_w Y_v} = \frac{0.25(1-e^{-0.0021 \times 1 \times 10^3})}{0.0021 \times 2.0} = 52 \text{ hm}^2 \text{ kg}^{-1} \quad (10)$$

$$\frac{B_v t_b}{P} = \frac{2.5 \times 10^{-3} \times 4.4 \times 10^5}{240} = 4.6 \times 10^0 \text{ hm}^2 \text{ kg}^{-1} \quad (11)$$

$$\begin{aligned} C_v &= 0.1 \{0.76(0.23) [52 + 4.6]\} + 0.9 \{0.76(0.23) (4.6)\} \\ &= 1.71 \text{ pCi/kg} \end{aligned} \quad (12)$$

And the resulting dose commitment for a population of 10,000 persons is:
 $1.71 \text{ pCi/kg} \times 2.0 \times 10^7 \text{ kg}^* \times 5.54 \times 10^{-8} \text{ rem/pCi} = 1.9 \text{ person-rem.}$

As in the dose commitments resulting from direct ingestion, the population dose commitments resulting from ingestion of contaminated crops are summed for all diversions along the model river.

C. Ingestion of Contaminated Beef

The 50-year population dose commitment resulting from ingestion of contaminated beef is the product of the dose factor (rem/pCi) times the beef activity (pCi/kg) times the total population intake (kg) for one year.

The beef activity, C_b , is calculated from the activities of the cattle drinking contaminated water and eating contaminated feed grain using the following equation (1):

$$C_b = F_b (C_w I_w + C_v I_v) \quad (13)$$

*Value from Table C-21, kg of vegetables per 10,000 people.

where C_w = activity in drinking water (pCi/l)

I_w = intake of drinking water (l/day)

C_v = activity in feed grain (pCi/kg)

I_v = intake of feed grain (kg/day)

F_b = beef transfer factor (day/kg).

$I_w = 50$ l/day, $I_v = 50$ kg/day and F_b for $Np-237 = 2.0 \times 10^{-4}$ (day/kg). (54)

The only difference between C_v for the human ingested crops and C_v for livestock ingested crops is that the vegetative yield (Y_v) used for the livestock crops is 0.75 kg/m^2 (1). This value gives a C_v of:

$$\frac{r(1-e^{-\lambda_w t_b})}{\lambda_w Y_v} = \frac{0.25(1-e^{-0.0021 \times 1 \times 10^3})}{0.0021 \times 0.75} = 139 \text{ hm}^2 \text{ kg}^{-1} \quad (14)$$

$$C_v = 0.1 \{0.76(0.23) (139 + 4.6)\} + 0.9 \{0.76(0.23) (4.6)\} \\ = 3.23 \text{ pCi/kg} \quad (15)$$

Thus, for the beef grown at the first diversion in River Model "A,"

$$C_b = 2.0 \times 10^{-4} \text{ day/kg} [(0.76 \text{ pCi/l} \times 50 \text{ l/day}) + (3.23 \text{ pCi/kg} \\ \times 50 \text{ kg/day})], = 0.04 \text{ pCi/kg} \quad (16)$$

For all beef pathways, a beef yield of 1 kg/100 kg feed crops was used (Cattle Marketing Information Service). This gives a total beef yield (population intake) = $3.9 \times 10^6 \text{ kg}$ for the sample calculation. The resulting dose commitment for a population of 10,000 persons is then:

$$0.04 \text{ pCi/kg} \times 3.9 \times 10^6 \text{ kg} \times 5.54 \times 10^{-8} \text{ rem/pCi} = 8.6 \times 10^{-3} \text{ person-rem.}$$

D. Ingestion of Contaminated Milk

The 50-year population dose commitment resulting from ingestion of contaminated milk involves a methodology identical to that for contaminated beef ingestion (see equation (13)). The parametric differences for the sample calculation are:

$$I_w = 60 \text{ l/day} \quad (54)$$

$$F_m = 5.0 \times 10^{-6} \text{ day/l} \quad (54)$$

*Value from Table C-21.

$$\begin{aligned}\text{Thus } C_m &= 5.0 \times 10^{-6} [(0.76 \text{ pCi/l} \times 60 \text{ l/day}) + (3.23 \text{ pCi/kg} \times 50 \text{ kg/day})] \\ &= 1 \times 10^{-3} \text{ pCi/l}\end{aligned}$$

The resulting dose commitment for a population of 10,000 people is then:

$$3.2 \times 10^7 \text{ l/yr} \times 1 \times 10^{-3} \text{ pCi/l} \times 5.54 \times 10^{-8} \text{ rem/pCi} = 1.8 \times 10^{-3} \text{ person-rem.}$$